Status of westslope cutthroat trout in the Madison River basin : the influence of dispersal barriers and stream temperature

by Matthew Robert Sloat

A thesis submitted in partial fulfillment Of the requirements for the degree of Master of Science In Fish and Wildlife Management
Montana State University

Abstract:

I evaluated the contemporary distribution of westslope cutthroat trout (WCT) in the Madison River basin in relation to fish dispersal barriers and stream temperatures. Species presence and abundance was determined by electrofishing sample sections at 0.8 km (0.5 mi) intervals in tributary streams. Fish dispersal barriers were identified by surveying entire stream lengths. I used continuously recording digital thermographs to record summer temperatures at 76 sites in first- to fourth-order streams and developed a predictive model of average daily stream temperatures using variables derived from geographic information system data layers and published climate records. Westslope cutthroat trout (>90% purity) were present in .17 of the 58 streams sampled (79 of 318 sites). Nonnative trout species (including Icnpwn hybridized populations of WCT with more than 10% introgression) were found in 133 sample sites. Estimated densities of WCT ranged from 3 to 40 fish >75 mm total length per 100 m of stream (mean- 21.9, SE=3.2). Most WCT populations occupied relatively short stream lengths (mean= 4.51 km, SE= 1.1) located above some type of fish dispersal barrier. Westslope cutthroat trout were associated with habitats where average and maximum daily stream temperatures ¦ generally remained below 12°C and 16°C, respectively. Stream temperatures were significantly colder at sites occupied by WCT than sites occupied by normative salmonids. A linear regression model including channel elevation, air temperature, channel slope, riparian forest cover, and drainage area explained approximately 76% of the variation in average daily stream temperatures. Because low population sizes and isolation place many WCT populations at risk of extinction, my study suggests that WCT populations should be expanded using genetically pure populations from within the Madison River basin to ensure their long term persistence. Where WCT and normative salmonids segregated without the influence of dispersal barriers, distribution boundaries were related to stream temperatures. Therefore, land management practices which alter natural thermal regimes should be avoided in these areas. The temperature model developed in this study provides resource managers with a cost effective tool for assessing westslope cutthroat trout habitat suitability over broad geographic areas.

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Matthew Robert Sloat

This thesis has been read by each member of the thesis committee and has been found to be satisfactory regarding content, English usage, format, citations, bibliographic style, and consistency, and is ready for submission to the College of Graduate Studies.

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ABSTRACT

I evaluated the contemporary distribution of westslope cutthroat trout (WCT) in the Madison River basin in relation to fish dispersal barriers and stream temperatures. Species presence and abundance was determined by electrofishing sample sections at 0.8 km (0.5 mi) intervals in tributary streams. Fish dispersal barriers were identified by surveying entire stream lengths. I used continuously recording digital thermographs to record summer temperatures at 76 sites in first- to fourth-order streams and developed a predictive model of average daily stream temperatures using variables derived from geographic information system data layers and published climate records. Westslope cutthroat trout (>90% purity) were present in 17 of the 58 streams sampled (79 of 318 sites). Nonnative trout species (including known hybridized populations of WCT with more than 10% introgression) were found in 133 sample sites. Estimated densities of WCT ranged from 3 to 40 fish >75 mm total length per 100 m of stream (mean= 21.9, SE=3.2). Most WCT populations occupied relatively short stream lengths (mean= 4.51 km, SE= 1.1) located above some type of fish dispersal barrier. Westslope cutthroat trout were associated with habitats where average and maximum daily stream temperatures generally remained below 12°C and 16°C, respectively. Stream temperatures were significantly colder at sites occupied by WCT than sites occupied by nonnative salmonids. A linear regression model including channel elevation, air temperature, channel slope, riparian forest cover, and drainage area explained approximately 76% of the variation in average daily stream temperatures. Because low population sizes and isolation place many WCT populations at risk of extinction, my study suggests that WCT populations should be expanded using genetically pure populations from within the Madison River basin to ensure their long term persistence. Where WCT and nonnative salmonids segregated without the influence of dispersal barriers, distribution boundaries were related to stream temperatures. Therefore, land management practices which alter natural thermal regimes should be avoided in these areas. The temperature model developed in this study provides resource managers with a cost effective tool for assessing westslope cutthroat trout habitat suitability over broad geographic areas.

INTRODUCTION

Because of their popularity as sport fish, many salmonid species have been transplanted outside their native ranges throughout North America. Introductions of nonnative salmonids have typically resulted in range constriction or elimination of native species through predation, competition, or hybridization (Gresswell 1988; Behnke 1992). In some situations native salmonids have persisted in the presence of introduced species, but the mechanisms that regulate displacement, and the habitat conditions that provide refuges for native species are not well understood (Fausch 1988; Gresswell 1988; Bozek and Hubert 1992).

As with other interior stocks of cutthroat trout *Oncorhynchus clarki*, populations of westslope cutthroat trout *O. c. lewisi* have declined throughout their historic range (Hanzel 1959; Liknes and Graham 1988; Behnke 1992). In Montana, declines of westslope cutthroat trout have been most dramatic within the Missouri River basin, with genetically pure populations occupying less than 5% of the subspecies' historical range (Shepard et al. 1997). The original distribution of westslope cutthroat trout within the Missouri River basin is thought to include the entire Missouri River drainage upstream from Fort Benton, Montana, including the Gallatin, Madison, and Jefferson drainages, as well as the headwaters of the Judith, Milk, and Marias rivers, which join the Missouri River downstream from Fort Benton (Behnke 1992). Within the Madison River drainage, westslope cutthroat trout abundance and distribution declined rapidly early in this century (USFWS 1999). Prior to about 1900, the Madison River and its principal tributaries supported abundant populations of westslope cutthroat trout upstream to barrier falls on the lower Firehole and Gibbon rivers in Yellowstone National Park (Jordan 1891). However, by the early 1950's westslope cutthroat trout no longer occurred in the Madison River or its principal tributaries within Yellowstone National Park (Benson et al. 1959), and were restricted to headwater habitats elsewhere in the drainage (Hanzel 1959).

Westslope cutthroat trout are listed as a Species of Special Concern by the Bureau of Land Management, a Sensitive Species by the U.S. Forest Service, and a Class A State Species of Special Concern by the Montana Department of Fish, Wildlife and Parks (MFWP) and the Montana Chapter of the American Fisheries Society (MFWP 1999). In 1997 westslope cutthroat trout were petitioned for listing as "threatened" under the Endangered Species Act. However, the U.S. Fish and Wildlife Service concluded that listing of this subspecies was not warranted (USFWS 2000). Current management efforts of state and federal agencies are aimed at locating and protecting remaining westslope cutthroat trout populations, expanding the distribution of genetically pure populations, and evaluating habitat conditions that provide refuge areas for remnant populations (MFWP 1999).

Factors responsible for the decline of westslope cutthroat trout include habitat alterations caused by land and water use practices, overharvest, and introductions of nonnative fishes (Hanzel 1959; Liknes and Graham 1988; Behnke 1992; McIntyre and Rieman 1995). Interactions with nonnative species through predation, competition, or hybridization probably constitute the greatest contemporary factor responsible for the loss of westslope cutthroat trout populations (Allendorf and Leary 1988; Liknes and Graham 1988; USFWS 1999). During the last 100 years, federal, state, and local agencies introduced several nonnative salmonid species into the Madison River (MFWP 2000). Rainbow trout *O. mykiss* and brown trout *Salmo trutta* were stocked periodically into the Madison River and its tributaries as early as 1889 (USFWS 1999) and were well established by the 1930's (USFWS 1954). Releases of hatchery-raised rainbow trout into the Madison River continued until 1974 (Vincent 1987). Yellowstone cutthroat trout *O. c. bouvieri* have been stocked in the Madison River drainage since the early 1950's, primarily in high mountain lakes of the Madison Range, but also in many stream habitats. Stocking of Yellowstone cutthroat trout within the Madison Range continued through the period of my study (MFWP 2000).

Extant populations of westslope cutthroat trout within the Madison River drainage are now restricted to headwater habitats, often above the upstream limit of nonnative salmonids (Sloat et al. 2000). In allopatry, westslope cutthroat trout are capable of inhabiting a much broader range of habitats. Populations of westslope cutthroat trout have experienced dramatic spatial and temporal variability in climatic and hydrologic conditions since the last glacial period (Behnke 1992). Before anthropogenic impacts dramatically reduced the range of westslope cutthroat trout, this species was found in a diverse array of habitats, which included small headwater streams and larger rivers, as well as mid- to low-elevation lakes (Shepard et al. 1984; Marnel 1988; Behnke 1992).

Additionally, individuals are known to make extensive migrations between these habitats (Bjornn and Mallet 1964; Shepard et al. 1984; Schmetterling, in press).

Interactive niche compression resulting from the presence of nonnative salmonids may partially explain the confinement of westslope cutthroat trout to headwater habitats (Mullan et al. 1992). Fausch (1989) hypothesized that colder, higher gradient headwater habitats provide refuges for cutthroat trout, where either more dominant nonnative salmonids cannot persist or where environmental conditions tip the balance of interspecific competition to favor cutthroat trout. Behnke (1992) suggested that cutthroat trout might have a selective advantage over nonnative trout in headwater areas because they may function better in cold environments. Field studies have demonstrated the importance of temperature in shaping cutthroat trout distribution (Mullan et al. 1996; Dunham et al. 1999). In laboratory experiments, temperature influenced competitive interactions between cutthroat trout and nonnative salmonids (DeStaso and Rahel 1994). Cutthroat trout also have slightly lower thermal tolerances than nonnative salmonids (Heath 1963; Feldmuth and Erikson 1978; DeStaso and Rahel 1994). Even relatively small differences in salmonid thermal tolerances can reflect substantial differences in growth optima (Takimi et al. 1997), competitive ability (DeStaso and Rahel 1994), and regional distributions (Fausch et al. 1994). Consequently, the influence of temperature on the distribution of westslope cutthroat trout has become a central concern in management for this subspecies. However, despite evidence that temperature is important, relatively little information is available to assess the thermal regimes that provide suitable habitat for westslope cutthroat trout or temperatures that provide refuges from competition and hybridization with introduced salmonids.

In this study, my goal was to explore how spatial patterns of fish dispersal barriers and stream temperature influenced the distribution of westslope cutthroat trout in the Madison River basin, Montana. Natural and anthropogenic dispersal barriers may restrict the distribution of salmonids (Kruse et al. 1997; Dunham et al. 1999) and in some cases, protect native salmonids from potential displacement by nonnative species (Rinne and Turner 1991; Young et al. 1996). Although generalized distributions of westslope cutthroat trout in Montana are known, the specific locations and characteristics of remnant populations within the Madison River basin were not available. Identifying and protecting existing populations is the first step in an effective conservation plan for westslope cutthroat trout (MFWP 1999). I assessed the distribution and abundance of westslope cutthroat trout and nonnative salmonids in tributaries to the Madison River between Hebgen and Ennis reservoirs.

Understanding fish-habitat relationships in streams has been a primary focus of fisheries biology. By identifying habitat conditions that limit fish distribution and abundance, biologists can focus management efforts on specific protection, enhancement, and mitigation practices to improve habitat conditions (Bozek and Rahel 1991). Because of evidence indicating the importance of temperature, I monitored summer thermal regimes in streams containing allopatric and parapatric westslope cutthroat trout populations, as well as streams where westslope cutthroat trout had been entirely displaced by nonnative salmonids. One reason for the paucity of information regarding the thermal influences on the distribution of westslope cutthroat trout is that many westslope cutthroat trout populations now exist primarily in headwater habitats. Often these habitats are in remote and rugged landscapes which make the acquisition of temperature data difficult, especially across broad geographic areas. When actual stream temperature data are not available, point elevations are commonly used as surrogates for stream temperatures (cf., Fausch et al. 1988). However, correlations between point elevations and stream temperatures are often weak (Isaak and Hubert 2001). Therefore, a predictive stream temperature model would provide a practical and cost-effective tool for resource managers working within mountainous landscapes where the remoteness and extent of the study area often preclude monitoring of thermal regimes throughout all drainages. Many factors operate to determine the thermal regimes of stream habitats (Ward 1985), but I hypothesized that temperatures of small streams were primarily controlled by climate and geomorphology. I tested this hypothesis using a series of multiple regression models that incorporated easily obtained landscape and climate data as causal variables, calibrated against observed summer stream temperatures in first- to fourth-order tributaries (Strahler 1957) of the Madison River.

My specific research objectives were to:

1. Describe the contemporary distribution and abundance of westslope cutthroat trout in the Madison River basin.
2. Determine the influence of fish dispersal barriers on the distribution and abundance of westslope cutthroat trout.

3. Determine the thermal characteristics of habitats occupied and unoccupied by westslope cutthroat trout.
4. Develop a predictive model of daily temperatures in small streams based on landscape variables estimated within a Geographic Information System (GIS) and published climate records.

STUDY AREA

This study took place in the 906 km^2 Madison River Valley, a north-trending intermontane basin located in southwest Montana. The Madison River is formed at the confluence of the Firehole and Gibbon rivers in Yellowstone National Park and flows approximately 195 km northward before joining the Gallatin and Jefferson rivers to form the Missouri River near the town of Three Forks, Montana. I sampled tributaries to the 101 km section of the Madison River between Hebgen and Ennis reservoirs. In this section, the Madison River flows through a broad valley flanked by large alluvial fans and fluvial terraces with a channel gradient of about 0.5% (USFWS 1954). Flow regimes in tributary streams are driven by snowmelt, and peak discharges occurred in May and June. Flows in the Madison River are regulated by Hebgen Dam, with peak discharges occurring in June and early July. The U.S. Geological Survey flow station below Hebgen Dam (station number 06038500) has operated since 1938. Flows recorded at this site were above average during the 1997, 1998 and 1999 water years (October through September) (Figure 1).

The Madison Range, which forms the eastern border of the study area, is within the Absaroka-Gallatin-Madison-Bridge Sedimentary Mountain ecoregion, a carbonate-rich, mostly forested, and partially glaciated region (Woods et al. 1999). The Madison Range rises dramatically from the valley floor to peak elevations exceeding 3,200 m. The Eastern Gravelly Mountain ecoregion forms the western border of the study area (Woods et al. 1999). The Gravelly Range is less rugged than the Madison range with

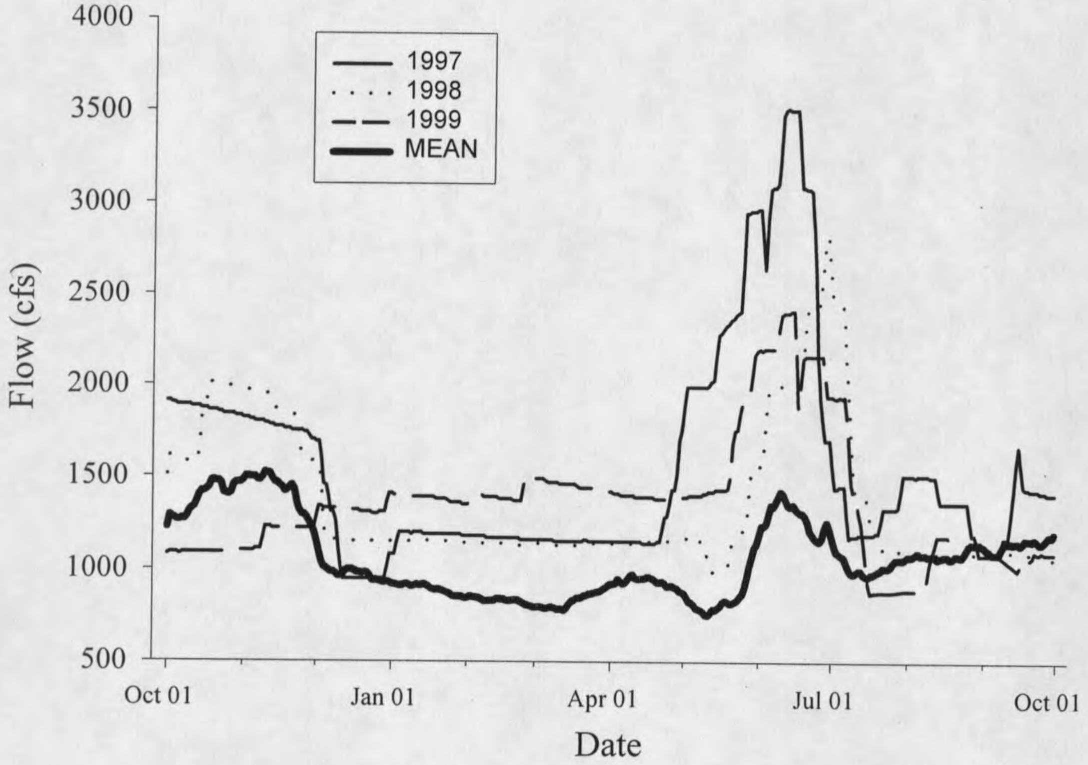


Figure 1. Daily flows in the Madison River below Hebgen Lake during this study (1997-1999) along with average daily flows for the 61-year period of record (1938-1999).

elevations not exceeding 2,900 m. The lower slopes of the Gravelly and Madison ranges and floor of the Madison River valley lie within the Dry Intermontane Sagebrush Basins ecoregion, which is composed of alluvium, fan, and valley fill deposits with natural vegetation of sagebrush steppe (Woods et al. 1999).

Although the alluvial plain in the Madison River valley is predominately privately owned, the majority of the basin is public and managed by the U.S. Forest Service. The primary land use in the Madison Valley is livestock grazing with localized dryland and irrigated agriculture. Limited logging has occurred on U.S. Forest Service land in the Gravelly Mountains. Land use is restricted in the Lee Metcalf Wilderness Area, which encompasses most of the Madison Range within my study area.

The climate of the Madison River Valley is typical of high-elevation intermontane basins with mild summers and cold winters. The average annual precipitation is 33.7 cm, and the average annual air temperature is 6.4°C on the valley floor (NOAA 1999).

Streams surveyed included Arasta, Buffalo, Hyde, English George, Quaking Aspen, Wall, Alpine, Tepee, Horse, Soap, Standard, and Wigwam creeks that drain the Gravelly Mountains; and Bear, Burger, Cabin, Corral, Cougar, Gorge, Indian, Manley, McAtee, Mill, Moose, No Man, Papoose, Raw Liver, Shedhorn, Shell, Squaw, Stock, Tolman, and Wolf creeks that drain the Madison Range (Figure 2). Streams ranged from first to fourth-order (measured from 1:24,000-scale USGS topographic maps after Strahler (1957)) with drainage areas between 9.2 and 128.8 km^2 (Table 1). Detailed descriptions of individual streams are provided in Sloat et al. (2000).

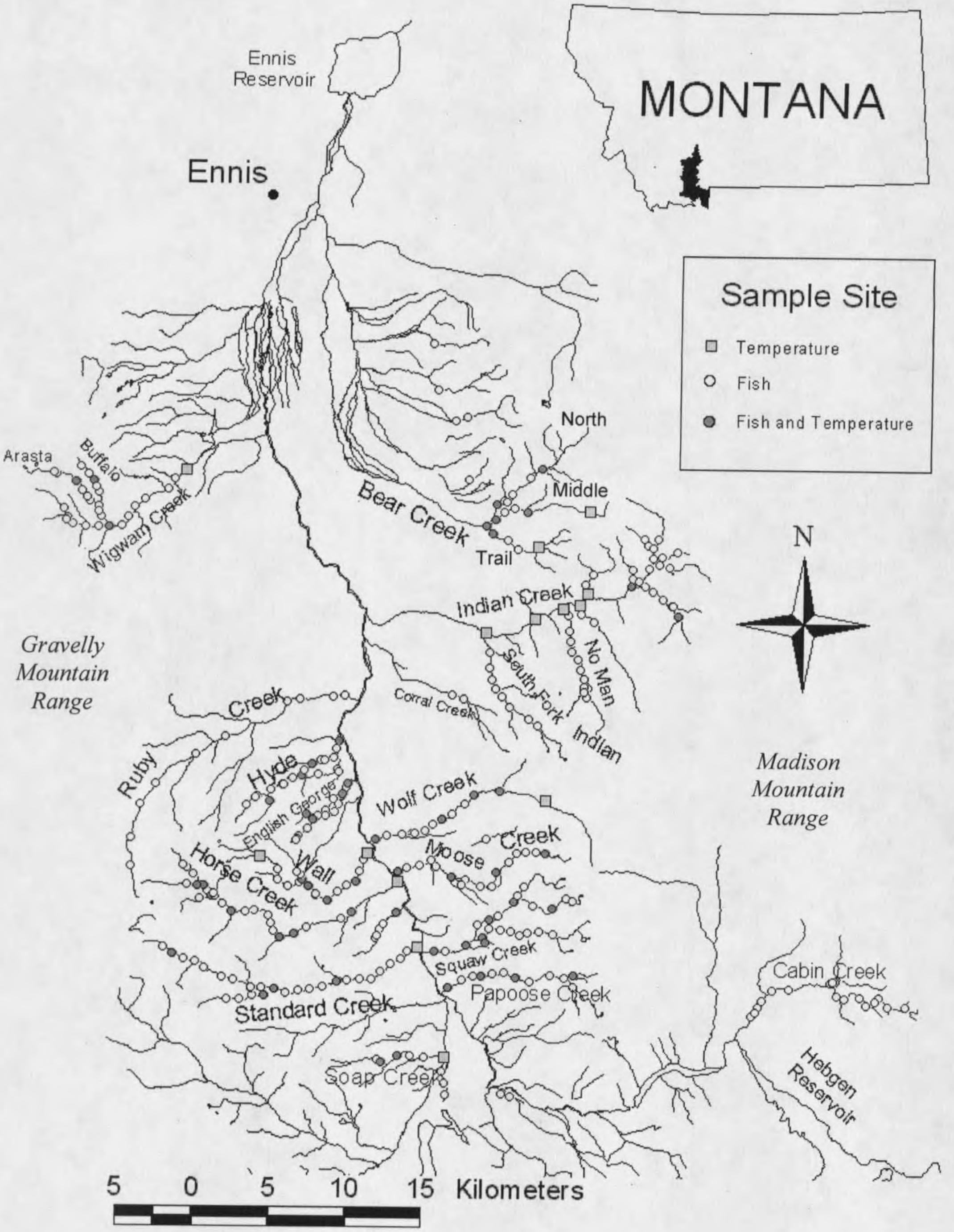


Figure 2. Map of Madison River drainage from Hebgen Reservoir to Ennis, Montana showing names of major streams sampled, and sample sites by type (Temp = temperature recording site; Fish = electrofishing sample sites; Fish and Temp = temperature recording and electrofishing sample site). Lower reaches of, Bear, Corral, Indian, and Wigwam creeks lacked surface flow.

Table 1. Watershed and stream characteristics for principal tributaries to the Madison River sampled in this study.

Stream name	Stream order	Drainage area (km^2)	Discharge (m^3/s)[1]		Stream length (km)
			July	September	
North Fork Bear	3rd	24.9	0.43	0.20	9.5
Middle Fork Bear	3rd	20.6	0.45	0.19	10.1
Trail Fork Bear	3rd	17.1	0.16	0.05	6.1
Indian	4th	128.8	3.06	0.92	26.5
Wolf	3rd	49.1	1.10	0.38	19.0
Moose	3rd	35.8	0.48	0.22	13.7
Squaw	3rd	44.6	1.76	0.38	14.2
Papoose	2nd	20.4	0.74	0.21	12.6
Wigwam	3rd	77.4	0.47	0.22	15.8
Hyde	2nd	18.5	0.06	0.03	6.3
English George	2nd	9.2	0.02	0.02	5.5
Wall	2nd	27.6	0.19	0.13	13.8
Horse	3rd	62.6	0.80	0.55	20.1
Standard	3rd	61.5	1.10	0.38	20.4
Soap	2nd	13.3	0.11	0.06	8.8
Ruby	3rd	83.8	-	-	24.8
Cabin	3rd	77.7	-	-	12.4

[1] Discharge measured from 7/13/99-7/20/99, and 9/18/99-9/30/99

Within the Madison River drainage the only native salmonids to co-occur with westslope cutthroat trout were mountain whitefish *Prosopium williamsoni* and arctic grayling *Thymallus arcticus*. The Madison River grayling population disappeared as early as 1920 (USFWS 1954), and only a vestigial population now inhabits Ennis Reservoir. Native non-salmonid fishes that occur in the Madison River drainage include white sucker *Catastomas commersoni*, longnose sucker *Catastomas catastomas*, mountain sucker *Catastomas platyrhynchus*, longnose dace *Rhinichthys cataractae*, and mottled sculpin *Cottus bairdi* (FERC 1997). Mottled sculpin was the only non-salmonid species sampled. Introduced salmonids sampled in this study were Yellowstone cutthroat trout, rainbow trout, and brown trout.

METHODS

Fish Distribution

A systematic sampling scheme was employed to estimate both the relative abundance and distribution of fishes. I attempted to survey as many streams as possible within the study area. However, because my primary objective was to locate remnant westslope cutthroat trout populations, I did not sample some streams if previous inventories conducted by MFWP indicated that they contained only nonnative species. I sampled streams at 0.8 km (0.5 mi) intervals by single-pass electrofishing with backpack Smith-Root electrofishers (Models SR-15B, SR-12B). At 3.2 km (2.0 mi) intervals, multiple-pass depletion population estimates were made (VanDeventer and Platts 1985). This protocol was modified slightly in some streams where more frequent sampling was done to document the upper and lower extent of the distribution of each fish species. Because of time constraints, this protocol was also modified in Squaw Creek where the stream was sampled at 1.6 km (1.0 mi) intervals, except where more frequent sampling was done to document the upper and lower extent of the distribution of each fish species. I was unable to backpack electrofish Indian Creek below its confluence with McAtee Creek (stream km 21.7) because of its large size. Hook-and-line sampling and angler interviews were conducted to verify fish species presence in this portion of Indian Creek, but no attempt was made to quantify fish densities. Sampling progressed upstream until trout were no longer present, then an additional upstream site was usually sampled to ensure fish absence. Exceptions occurred when upper limits to fish distributions were associated with minimal stream flows. Total length (TL) and weight were recorded for all captured salmonids. Sampling of fish distributions was conducted primarily during the summers of 1997, 1998, and 1999, although fish distribution data for Soap Creek were collected in 1995.

When conducting multiple depletion population estimates, if field calculated probabilities of capture (calculated as 1- [C2/C1]; where C1= number captured on the first pass, and C2 = number captured on second pass) were less than 0.80 after two passes, additional electrofishing passes were made (cf., Riley and Fausch 1992). Relative fish abundance was calculated by species as the number of fish 75 mm TL or longer per 100 m of stream captured in the first electrofishing pass for all sampling events. Population estimates were calculated using a maximum likelihood estimator within the MICROFISH program (Van Deventer and Platts 1985) by species for fish 75 mm TL or longer and standardized as the number of fish per 100 m of stream length.

To help insure that all species present were included in the sample, sample section lengths were at least 35 times the average wetted stream width (Lyons 1992). Sample sites were referenced by kilometer above the stream's mouth and by latitude and longitude obtained from a Global Positioning System (GPS). Sample site locations were input into an ArcView (ESRI 1999) event theme and projected on 1:100,000 stream hydrography layers. The field GPS locations were corrected to overlay the hydrography layer and stream kilometer locations when discrepancies existed between field GPS and mapped locations.

Either whole fish or fin samples from westslope cutthroat trout were taken for genetic analysis. Genetic characteristics were determined by horizontal starch gel electrophoresis (whole fish) or by Paired Interspersed Nuclear DNA Element-PCR (PINE [fin clips]) by the University of Montana Wild Trout and Salmon Genetics Laboratory.. Where possible, fin clips were taken from 25 westslope cutthroat trout per stream. With a sample size of 25 fish, there is a 95% chance of detecting as little as a 1% Yellowstone cutthroat or rainbow trout genetic contribution to a hybridized population of westslope cutthroat trout (Spruell and Miller 1999). Where possible, a sub-sample of westslope cutthroat trout captured at each sample site within a stream was represented in the genetic analysis to detect longitudinal changes in genetic composition within a population.

Fish species identification was based on genetic testing results whenever possible. Fish were considered westslope cutthroat trout if frequencies of alleles characteristic of westslope cutthroat trout were 90% or higher (MFWP 1999). Known hybridized populations of westslope cutthroat trout with more than 10% introgression were classified as nonnative salmonids. When genetic test results were unavailable for a population, I used field identifications based on spotting pattern, body color, and presence/absence of an orange “cutthroat” slash below the lower mandible..

All potential barriers to fish movement were referenced by latitude and longitude using a hand-held GPS unit. Dispersal barriers were defined as structures with vertical drops at least 1.5 m high (Stuber et al. 1988; Kruse et al. 1997). Barrier locations were input into an ArcView event theme and projected on 1:100,000 stream hydrography layers. Natural physical dispersal barriers consisted of geologic waterfalls, and decadent beaver *Castor canadensis* dam complexes. Human-caused dispersal barriers consisted of irrigation diversion dams.

The length of habitat occupied by westslope cutthroat trout and nonnative salmonids was calculated for each tributary drainage and then compared using Welch's modified t-test which does not assume equality of group variances (Zar 1984). Occupied habitat lengths were defined as the total occupied stream kilometers in a drainage not interrupted by a dispersal barrier, and did not include the main stem of the Madison River.

Stream Temperature

Continuously recording digital thermographs ("Hobo" and "Stowaway" models, Onset Corp.; http://www.onsetcomp.com) were used to record water temperatures in first- to fourth-order streams (Strahler 1957) across the Madison River basin (Figure 3). Thermographs were capable of measuring temperatures ranging from -5° C to 37° C with an accuracy of +/- 0.2°C and +/- 0.14 min/day. Prior to field deployment, thermographs were calibrated against a National Institute of Science and Technology hand held thermometer at 3, 9, and 20°C to ensure their reliability. All the thermographs tested measured water temperatures within the accuracy reported by the manufacturer.

During all years, thermographs were placed in streams during early July and left to record stream temperatures until late September. In streams where the distributions of trout species were known *a priori*, thermographs were placed at upper and lower distribution boundaries. However, for many streams distribution boundaries were not

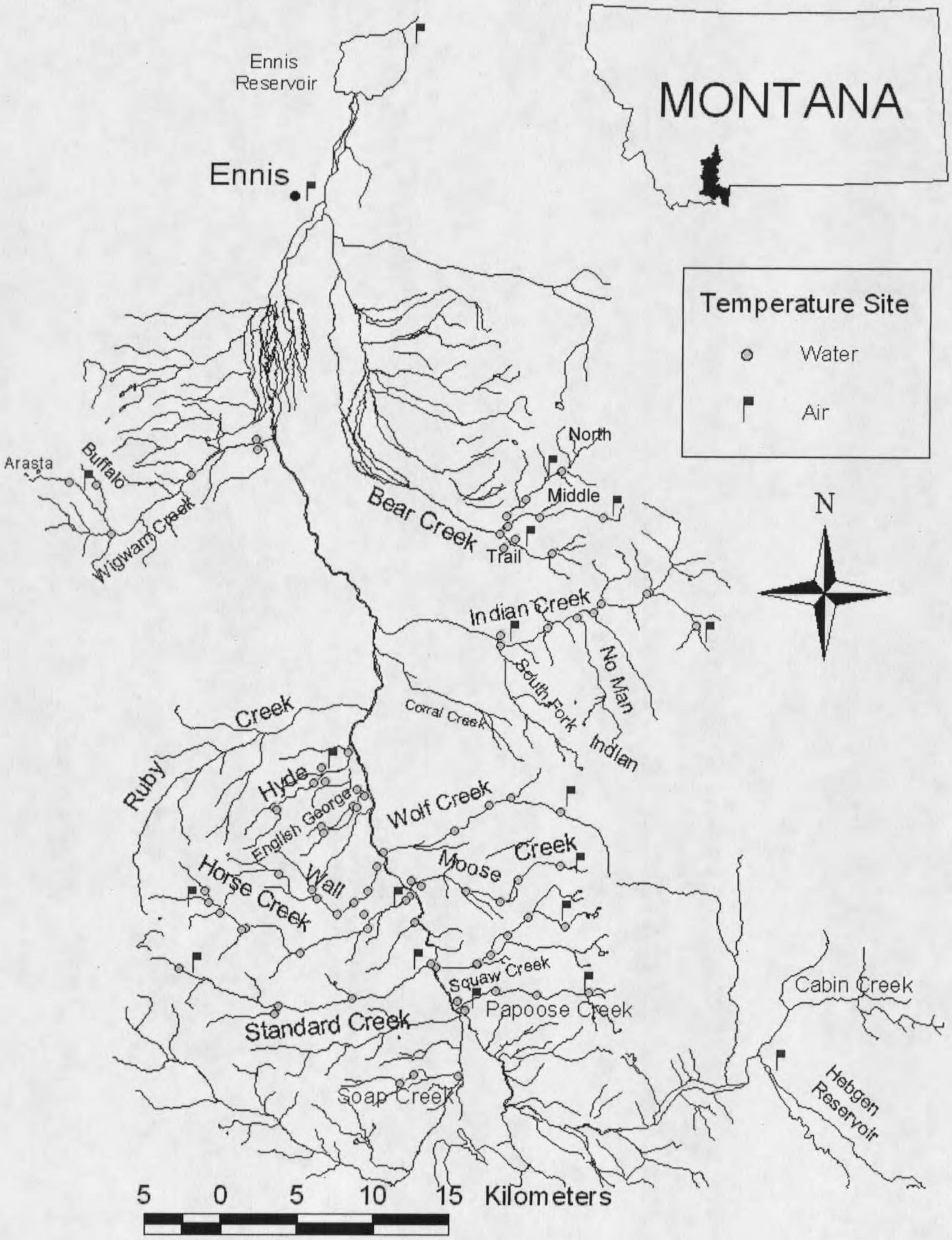


Figure 3. Location of air and water thermograph recorders placed in Madison River tributaries from 1997 to 1999.

known prior to thermograph deployment. In these streams thermographs were uniformly distributed along the stream's length. A minimum of 3 thermographs were used per sub-drainage and were placed from 1 to 7 km apart in principal study streams as well as at the mouths of smaller tributaries. Thermographs were placed in well-mixed run or pool habitats and were shielded from direct solar radiation. Thermographs recorded stream temperatures at 1 or 2 h intervals, depending on the memory capacity of the thermograph model. Hourly and bi-hourly stream temperatures were summarized into daily maxima, minima, and means. Because of a limited number of thermographs available for this study and the extensive time involved in placing thermographs in the field, not all tributaries sampled for fish had thermographs. One stream (Ruby Creek) was excluded from temperature sampling in order to sample other streams where nonnative salmonids occupied a greater altitudinal range. Additionally, the remote locality and lack of *a priori* knowledge of fish distribution and genetic status precluded thermograph placement in South Fork Indian and Cabin creeks.

Stream temperature data were collected from 76 sites in 1999 (Figure 3). However, at five locations thermographs were dried or otherwise affected by irrigation withdrawal for part of the summer; consequently, they were excluded from analysis. Stream temperatures were also measured at six sites during 1998, and two sites in 1997 (Figure 3).

The four most common methods of quantifying maximum stream temperatures, as well as degree days (Table 2), were examined for sample sites where both fish abundance and stream temperature were collected. During 1999, not all thermographs were in place by July 1st. Therefore, to facilitate comparison, degree-days were calculated from July 8–September 15 when all thermographs were in place and recording. I used t-tests ($\alpha = 0.05$) to test the hypothesis that stream temperatures were significantly colder at sites occupied by westslope cutthroat trout than those occupied only by nonnative salmonids. Fish sampling events were matched with temperature records corresponding most closely in time, but in some cases stream temperatures were not measured during the same year as the fish sampling event. For this analysis, I made two assumptions: 1) fish distribution boundaries did not change over the relatively short time period of my study, and 2) measured stream temperatures were representative of temperatures experienced by fish during the year fish distribution data were collected. Published air temperature records for the period of my study indicate that annual and summer air temperatures at Ennis, Montana varied less than 1°C from long term average air temperatures (period of record 1948-1999, NOAA 1999). Because extremely low densities of fish at some locations may have biased the analysis by including sample sites with marginal habitat, I did not include sites if fish densities were less than 3 fish >75 mm TL per 100 m of stream.

Table 2. Definition of temperature metrics used in this study.

Metric	Abbreviation	Definition
Degree Days	DD	Sum of average daily temperatures over 0°C
Maximum average daily temperature	MDAT	Maximum of average daily temperature within a year
Maximum daily maximum temperature	MDMT	Maximum of maximum daily temperature within a year
Maximum weekly average temperature	MWAT	Maximum seven-day average of daily average water temperatures
Maximum weekly maximum temperature	MWMT	Maximum seven-day average of daily maximum water temperatures

Temperature Modeling

Multiple regression models were developed for Madison River basin streams with average daily water temperature as the response variable. The models were based on studies showing stream water temperature to be most sensitive to air temperature (Stephan and Preud'homme 1993), solar radiation (Bartholow 1989; Stephan and Preud'homme 1993), and catchment geomorphology (Smith and Lavis 1975; Isaak and Hubert 2001). Because I was primarily interested in whether stream temperatures could be estimated from variables estimated within a GIS, variable selection was also guided by the availability of GIS coverages to provide data for estimating those variables. Daily air temperatures, channel elevation, channel aspect, channel slope, land cover, and stream size were included as possible predictor variables.

All variables, except air temperature, were quantified with the aid of a GIS that comprised several coverages for the Madison River drainage. Coverages included a 30 m resolution digital elevation model (DEM), a 1:100,000-scale hydrology layer, point coverages of thermograph locations, and a 90 m resolution grid of land cover types. Completed coverages were acquired from the Beaverhead-Deerlodge National Forest, MFWP, and the Montana Gap Analysis Project (Redmond et al. 1998).

In order to characterize each sampling site, individual drainage boundaries were determined by delineating a “pourpoint” on the DEM at the thermograph location and using the Surface Water Modeler extension in ArcView to automatically delineate the watershed upstream from the pourpoint. Variables were then quantified for the resulting sub-drainages.

Elevation was quantified as both a point elevation of the thermograph location and as the mean elevation of the entire stream channel upstream from the thermograph. To calculate channel elevation, the hydrology data layer was converted to a 30 m grid and intersected with a DEM grid for the study area. The resulting grid was then analyzed using the “summarize zones” function in the Spatial Analyst extension in ArcView. Mean channel gradient was calculated for the entire stream network above each thermograph site from a slope coverage derived from the DEM in ArcView. Aspect was defined as the down-slope direction (the maximum rate of change in elevation along the stream channel) from each cell to its neighboring cells, expressed in positive degrees from 0 to 360, measured clockwise from the north. Numeric ranks (1-4) indicative of the relative solar radiation the drainage likely received were then assigned to compass directions (Shepard et al. 1998; Figure 4).

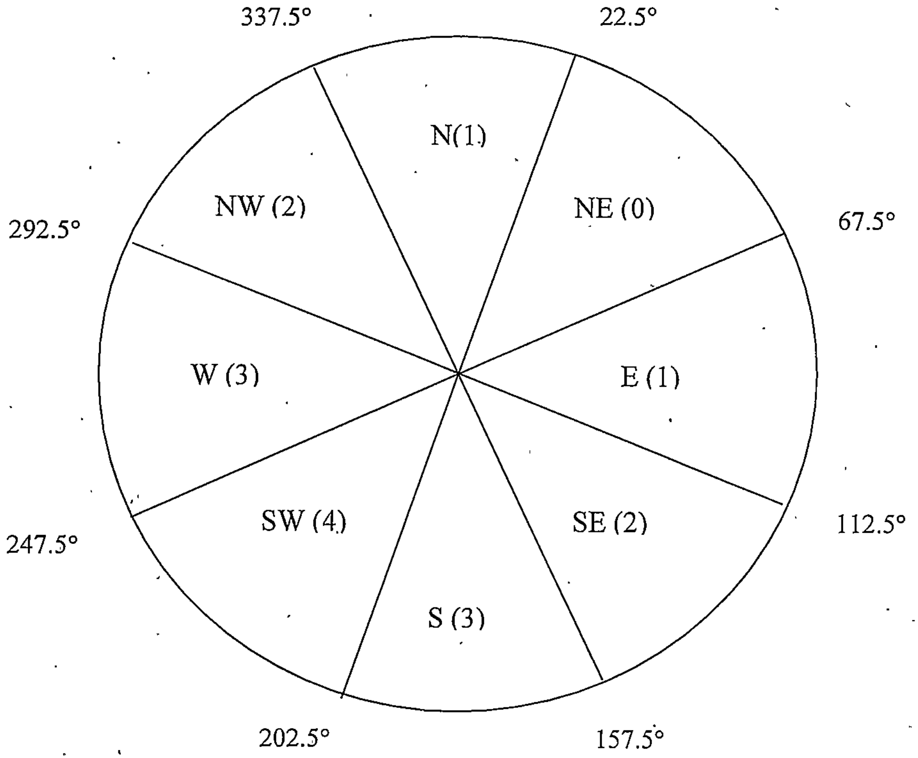


Figure 4. Criteria used to rank stream channel aspect for deriving integrated channel aspect above each sample site. Aspect compass directions and numeric ranks (in parentheses) assigned to each compass direction are shown. Numeric ranks indicate relative solar radiation the drainage likely receives.

Drainage area was calculated for the portion of the watershed above each thermograph site in ArcView. To identify relationships between drainage size and stream size, I also measured stream discharge at a subset of 25 temperature recording sites during mid-July, mid-August, and mid-September, 1999. Discharge was measured in each principal tributary to the Madison River at the lowest thermograph site not affected by irrigation withdrawal using a top-set rod and a digital flow meter. Additionally, discharge was measured at all temperature recording sites in Hyde, Wall, Standard, Wolf, and North Fork Bear creeks. Wetted channel widths were also determined for all thermograph sites as the average of 1 to 5 width measurements made within 100 m of the thermograph location at the time of thermograph placement. I then correlated drainage area calculations with on-site measurements of stream discharge and wetted stream width.

A 90 m grid of land cover types obtained from the Montana Gap Analysis Project (Redmond et al. 1998) was used to determine the influence of vegetative cover on stream temperatures. The land cover data set was composed of 45 cover types based on remote sensed imagery collected from 1991 to 1993. Estimated mean accuracy of cover type classification for the state of Montana was 61.4%, but exceeded 80% within my study area (Redmond et al. 1998). A recent study conducted in neighboring Gallatin County, Montana found overall classification accuracy of landcover types to be 88% (Oechsli 2000). Because I was primarily interested in shading effects of riparian vegetation, I reclassified vegetation cover types into either "tree" or "non-tree" categories. I applied a 200 m buffer centered on the stream channel and tabulated the percentage of tree cover within the buffer using the "summarize zones" function in the Spatial Analyst extension in ArcView. To determine the spatial extent of the influence of riparian shading on stream temperatures I used the percentage of tree cover within 100, 500, and 1000 m reaches located directly above each thermograph site, as well as along the entire stream network above a thermograph location.

Because ponds and lakes may dramatically influence the thermal regime of a stream, I also calculated the area of standing water connected to the stream channel within each sub-drainage, as well as the distance from a thermograph site to the lake outlet using the land cover grid and the 1:100,000 hydrology layer.

Since I was primarily interested in whether stream temperatures could be estimated using existing climate data sources, I matched daily maximum, minimum, and average air temperatures from published air temperature records with daily stream temperature records. However, because relationships between air and water temperatures determined by statistical regression may have been influenced by the elevation of the recording station, I used a variety of air temperature data sources to determine this influence. Continuously recording digital thermographs (Onset Corp.: http://www.onsetcomp.com) were used to measure air temperature at 16 sites throughout the Madison River drainage during 1999 (Figure 3). Air temperature recording thermographs were placed near the upper and lower ends of tributaries. Because of a limited number of thermographs, air temperatures in some sub-drainages were not measured in order to sample other sub-drainages with greater altitudinal ranges. Air temperature data were also obtained from published National Oceanic and Atmospheric Administration (NOAA) climatological records for five weather stations in or around the Madison Valley (NOAA stations = Alder, Ennis, Hebgen Dam, Norris-Madison Pump House, and Virginia City). Daily air temperature observations from different localities were systematically substituted into the regression model and used to predict stream temperatures at all sites. I then regressed coefficients from the resulting regression against the elevation of the air temperature data source to determine if any linear relationship existed between coefficient values and station elevation. This analysis was conducted to determine how to adjust the model coefficients if a new air temperature data source was used for predictions.

Best-subsets regression was used to select variables to be used in regressions. I used a variety of model selection criteria, including Akaike's information critierion (AIC), and adjusted coefficients of determination (adjusted R^2) for selection of alternative regression models (Neter et al. 1996; Burnham and Anderson 1998). I used Pearson's correlations, and variance inflation factors to identify relationships among GIS-derived landscape variables. Since identical air temperature values were used to estimate stream temperatures at all sites, air temperature was excluded from variable correlations. Model performance was assessed using a jackknife procedure which did not require the collection of an independent set of reference data. Each stream thermograph site was systematically excluded from the data set and the regression was reconstructed. The new coefficients were then used to predict temperatures for the site that was removed. This avoids using observations to simultaneously create and evaluate the predictive performance of the regression model. The Durbin-Watson test for autocorrelation of error terms was used to evaluate the influence of the time series nature of the data (Neter et al. 1996).

I tested for the effects of sub-drainage and sample year on average daily stream temperatures using the mixed procedure in SAS (SAS for Windows version 7.0; 1999). A mixed regression model was created using the variables from the "best" linear regression model entered as fixed variables, and sub-drainage and year as random variables. I used the Wald statistic to test for significance of random effects (Littel et al. 1996). These analyses were conducted to determine if there was additional correlation between observations that had the same level of either sub-drainage or year. For example, by entering sub-drainage as a random effect in the mixed regression model, I wanted to determine whether stream temperatures at a site within a sub-drainage had stronger correlations with temperatures at other sites in the same sub-drainage than with temperatures at sites in other drainages.

RESULTS

Fish Distribution

The distribution of westslope cutthroat trout in the Madison River drainage between Hebgen and Ennis reservoirs was determined using samples from 318 locations in 58 streams within 18 different sub-drainages (Figure 5). Westslope cutthroat trout (>90% purity; Appendix A) were present in 17, or 29%, of the 58 streams sampled (79 of 318 sites). Nonnative trout species, including rainbow, brown, and Yellowstone cutthroat trout, as well as hybridized westslope cutthroat trout with more than 10% introgression (Appendix A), were found in 133 sample sites (Figure 5). Hybrid cutthroat trout were present in 48 of the 133 sample sites occupied by nonnative salmonids. No fish were captured in 106 sample sites (Figure 5). Hook-and-line sampling and angler interviews confirmed the presence of rainbow trout throughout Indian Creek. However, I did not attempt to quantify fish densities in Indian Creek between South Fork Indian (stream km 9.6) and McAtee creeks (stream mile 21.7).

Within the Madison River basin, the distribution of westslope cutthroat trout was concentrated in streams draining the Gravelly Mountain Range. In this range, all westslope cutthroat trout populations but one were isolated from nonnative species by dispersal barriers. Natural barriers to fish dispersal were found in 8 of the 9 sub-drainages sampled in the Gravelly Range. In six sub-drainages, nonnative salmonids were present up to the base of the barrier and only westslope cutthroat trout were present

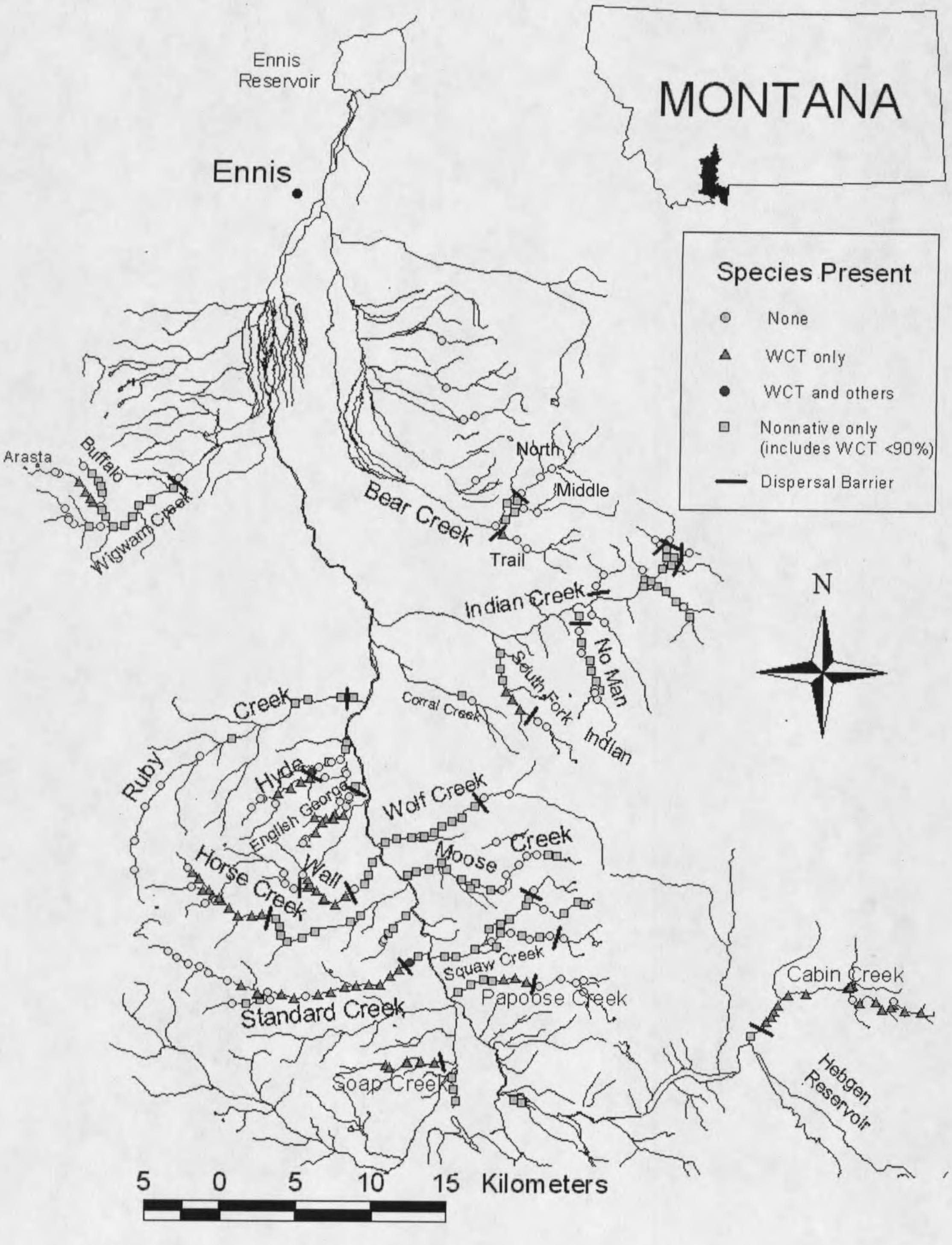


Figure 5. Map of upper Madison River drainage showing the distribution of westslope cutthroat trout and nonnative salmonids. Fish dispersal barriers corresponding with species distribution boundaries are also shown.

upstream. Because some sub-drainages included more than one occupied stream, these six sub-drainages represent occurrence in 11 streams. Nonnative salmonids were present both above and below barriers (2-5 m high vertical falls) to fish migration in the two remaining sub-drainages (3 streams).

In the Wigwam Creek sub-drainage, which included Arasta, Buffalo, and Wigwam creeks, westslope cutthroat trout occurred in Arasta Creek but were not isolated from rainbow x cutthroat hybrids inhabiting the remaining portion of the drainage by a physical dispersal barrier (Figure 5). Genetic sampling conducted in 1995 by MFWP indicated that Arasta and Buffalo creeks supported genetically pure westslope cutthroat trout (Appendix A). However, sampling done during this study indicated that fish in the upper Wigwam Creek sub-drainage (including Arasta and Buffalo creeks; see Figure 5), had greater than 10% introgression with both rainbow and Yellowstone cutthroat trout. Results from Arasta Creek were based on a single individual captured near the stream's mouth. Consequently, additional sampling is necessary to determine if upper Arasta Creek still supports a pure population of westslope cutthroat trout.

Genetic analysis of fish from Standard Creek also produced equivocal results (Appendix A). Morphologically, fish captured in Standard Creek resembled westslope cutthroat trout. Genetic analysis revealed that while westslope cutthroat trout genes were dominant in Standard Creek, some Yellowstone cutthroat trout introgression occurred. Additionally, a freezer malfunction made it impossible to detect the presence of rainbow trout alleles. Consequently, the extent of introgression with either Yellowstone cutthroat or rainbow trout could not be determined. Because of a lack of evidence to the contrary, for the purpose of this study, the Arasta and Standard creek populations were considered westslope cutthroat trout.

Except in Hyde Creek, where a large beaver dam complex prohibited upstream migration of nonnative salmonids, all dispersal barriers in streams draining the Gravelly Range consisted of geologic falls. Typically fish dispersal barriers were located relatively low in streams draining the Gravelly Range. The average distance above the stream's mouth and mean elevation of dispersal barriers in the Gravelly Range were 3.9 km (SE= 1.7) and 1,954 m (SE = 79), respectively.

Westslope cutthroat trout were found less frequently in streams draining the Madison Range than in the Gravelly Range. Only four of the 10 sub-drainages (6 of 35 streams) sampled in the Madison Range supported westslope cutthroat trout (Figure 5). In contrast to distribution patterns in the Gravelly range, only one westslope cutthroat trout population (Cabin Creek; Figure 5) was found above a natural fish migration barrier. Geologic dispersal barriers were found on 10 of the 33 streams supporting fish in the Madison Range (Figure 5). In eight of these streams, fish were present up to the base of the barrier and absent upstream. Introduced Yellowstone cutthroat trout were found above dispersal barriers in No Man Creek which contained a headwater lake regularly stocked by MFWP (MFWP 2000). Additionally, a very small population (<50 individuals) of westslope cutthroat trout was isolated above an irrigation diversion dam in Trail Fork Bear Creek. Fish dispersal barriers were located relatively far from stream mouths (mean= 12.5 km, SE= 2.9), and at high elevations in the Madison Range (mean=

2267 m, SE= 62), and occurred significantly farther upstream and at higher elevations than in the Gravelly Range (t-tests, $P<0.05$).

Basin-wide, westslope cutthroat trout were sympatric with nonnative trout in only two sample sites (Figure 5). Sites where westslope cutthroat trout occurred with nonnative species were located directly below barriers which protected upstream westslope cutthroat trout populations and few (2-3) westslope cutthroat trout individuals were captured at these locations (Appendix B). This pattern reflects the general correspondence of distribution boundaries with dispersal barriers, but in some cases, may have been influenced by genetic sampling methods. Because genetic proportions were determined from a sub-sample of fish at each sample site, and because sample sizes per site were low (range= 1-6 samples), some pure westslope cutthroat trout may have been present in stream reaches containing rainbow x cutthroat hybrid trout but not detected.

Relative abundances of westslope cutthroat trout captured during a single electrofishing pass ranged from 1 to 40 fish per 100 m of stream length (mean= 10.8, SE= 1, n= 79; Appendix B), compared to a range of 1 to 84 for nonnative salmonids (mean= 9.7, SE= 1, n=133; Appendix B). Mean relative abundance of westslope cutthroat trout was not significantly different from nonnative salmonid species (t-test, $P=0.43$). Estimated densities of westslope cutthroat trout from multiple depletion estimators ranged from 3 to 40 fish per 100 m of stream length (mean= 21.9, SE=3.2, n=20), compared to densities of 1 to 185 for nonnative salmonids (mean= 25.6, SE=7.8, n=34; Appendix C). Similar to findings for relative abundances, densities of westslope cutthroat trout were not significantly different from densities of nonnative salmonids based on depletion estimates (t-test, *P*= 0.66). Based on capture probabilities derived from multiple depletion estimates, the efficiency of single-pass removals was approximately 80% for all species combined and was slightly higher for nonnative salmonids (82%) than westslope cutthroat trout (79%), but this difference was not significant (t-test; *P*=0.79).

Despite their association with dispersal barriers, the length of habitat occupied by westslope cutthroat trout per sub-drainage (mean= 4.51 km, SE= 1.1) was not significantly different than that occupied by nonnative salmonids (mean= 4.99 km, SE= 1.2; t-test, *P*=0.77). However, when occupied habitat lengths for only westslope cutthroat trout populations were compared, isolated westslope cutthroat trout populations occupied longer stream lengths than did populations not isolated by fish barriers. Isolated westslope cutthroat trout occupied an average stream length of 7.5 km (SE= 2.2), while all non-isolated populations occupied approximately 2.4 km of stream.

Westslope cutthroat trout also reached greater abundances at sites above dispersal barriers (Figure 6). Mean westslope cutthroat trout abundance at sites above physical dispersal barriers was 12.8 fish per 100 m (SE= 1.1) compared to 3.8 fish per 100m (SE= 0.8) at sites not influenced by physical dispersal barriers, and this difference was statistically significant (t-test, *P*<0.001). This difference in fish abundance did not appear to be a function of limited physical habitat (estimated by wetted stream width), or sample site elevation (Figure 6).

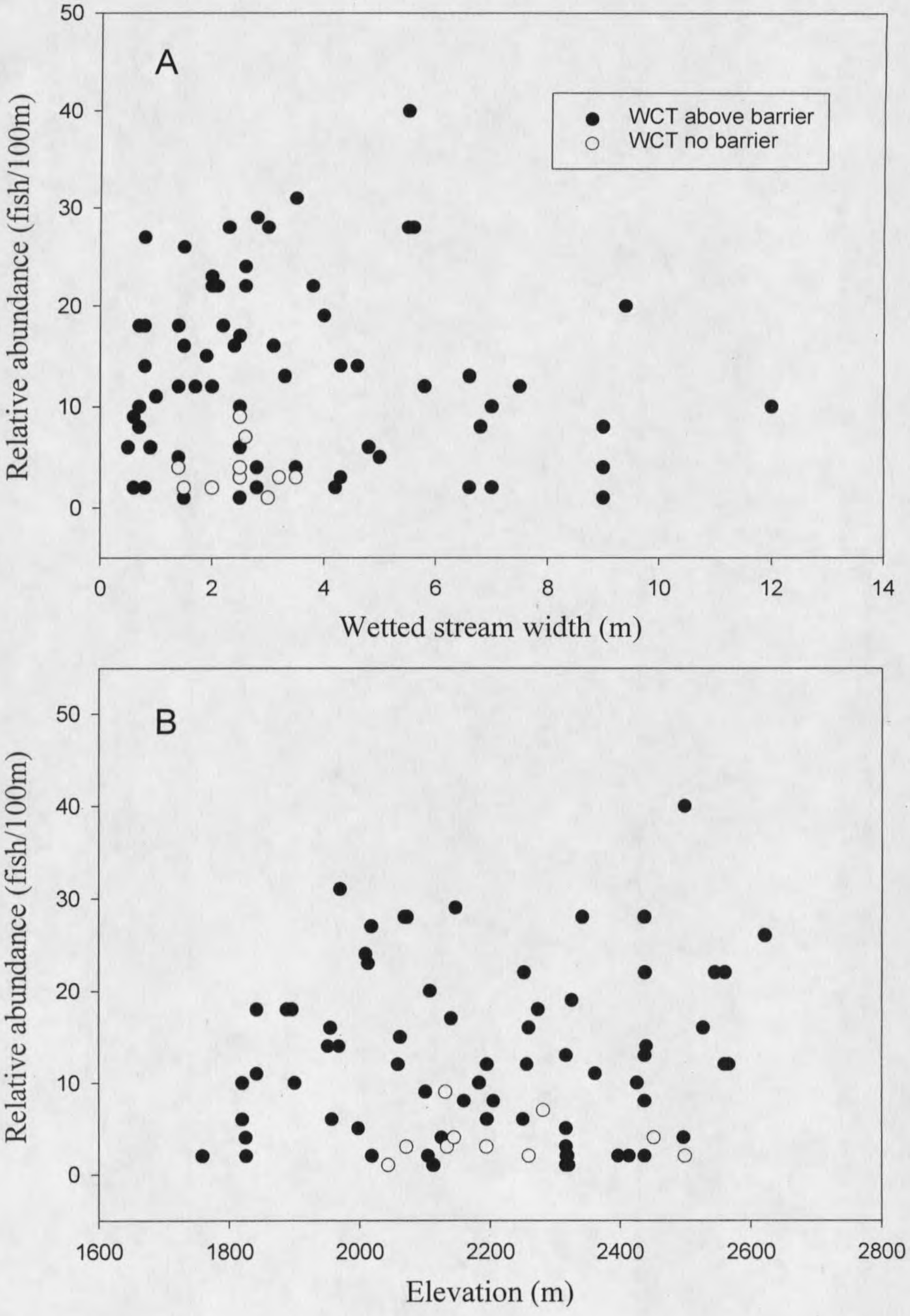


Figure 6. Relative abundance of westslope cutthroat trout (WCT) greater than 75 mm at sites without fish dispersal barriers (open circles) and at sites above fish dispersal barriers (filled circles) in relation to stream width (A) and elevation (B).

Stream Temperature

Westslope cutthroat trout were associated with habitats where average and maximum daily stream temperatures generally remained below 12°C and 16°C, respectively (July 1-September 15; Figures 7 and 8). Maximum daily average temperatures (MDAT) ranged from 7.2 to 12.7°C, and maximum daily maximum temperatures (MDMT) ranged from 9.9 to 16.5°C at sites occupied by westslope cutthroat trout during the summer sampling period (Table 3). Thermal regimes differed significantly between sites occupied by westslope cutthroat trout and nonnative salmonids. Although there was considerable overlap, stream temperature metrics MDAT, MWAT, MWMT, as well as degree days (DD), were significantly lower at sites occupied by westslope cutthroat trout than sites occupied solely by nonnative salmonids (t-tests, α=0.05; Table 3). Differences in MMDT at sites occupied by westslope cutthroat trout and sites occupied by nonnative salmonids were marginally significant (Table 3).

Table 3. Mean and range of five temperature metrics (see Table 2 for definitions) at sites occupied by westslope cutthroat trout (WCT) and sites occupied by nonnative trout species.

Temperature metric	WCT	Nonnative trout	*P*-value[1]
MDAT	9.84 (7.2-12.7)	11.08 (8.1-16.3)	0.033
MDMT	13.18 (9.9-16.5)	14.54 (10.6-22.0)	0.050
MWAT	9.47 (7.1-11.7)	10.6 (7.8-15.1)	0.022
MWMT	12.26 (9.3-15.3)	13.79 (10.0-23.1)	0.027
DD	563.9 (414.5-693.2)	626.7 (465.7-882.6)	0.030

[1] Welch's modified t-test.

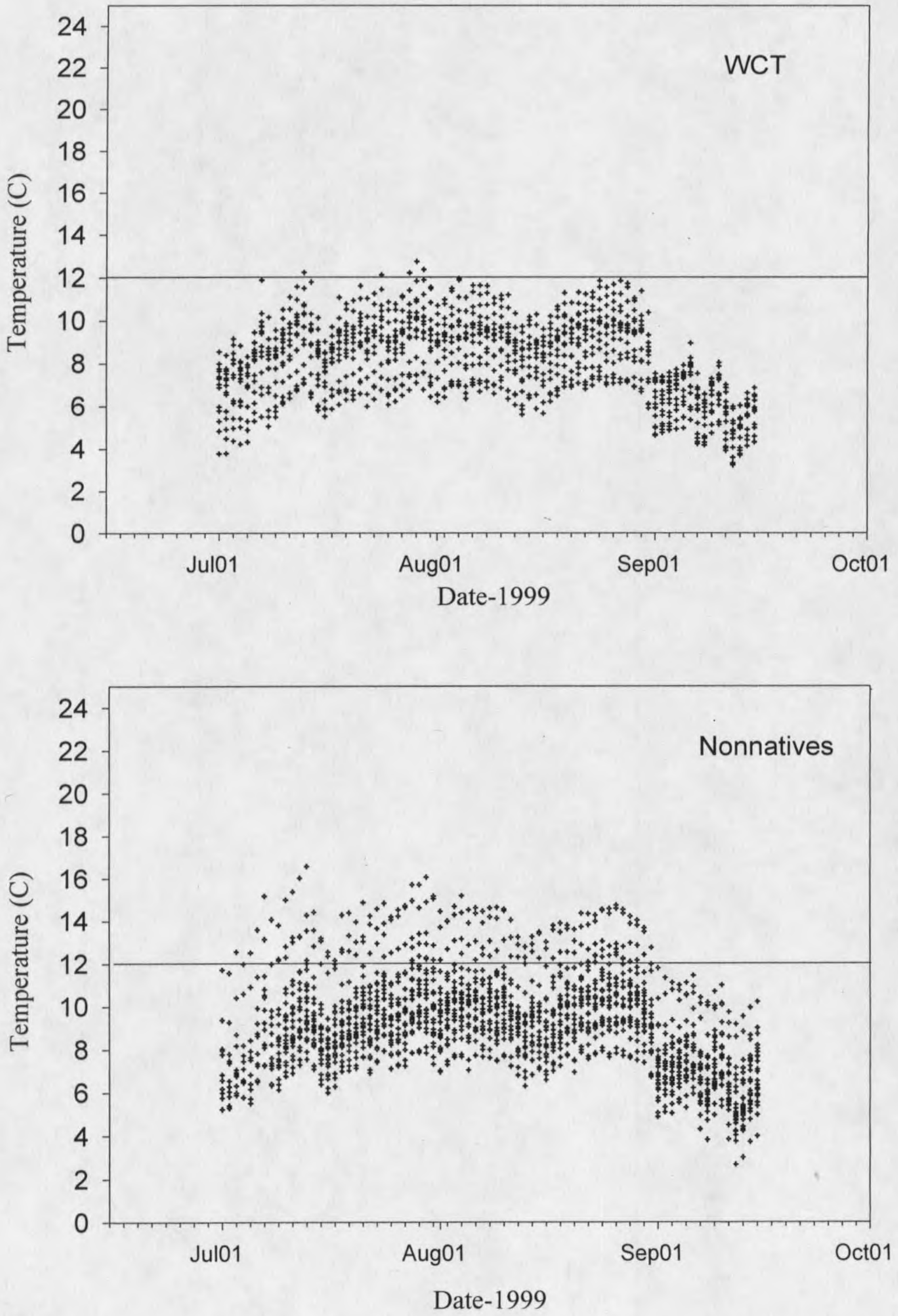
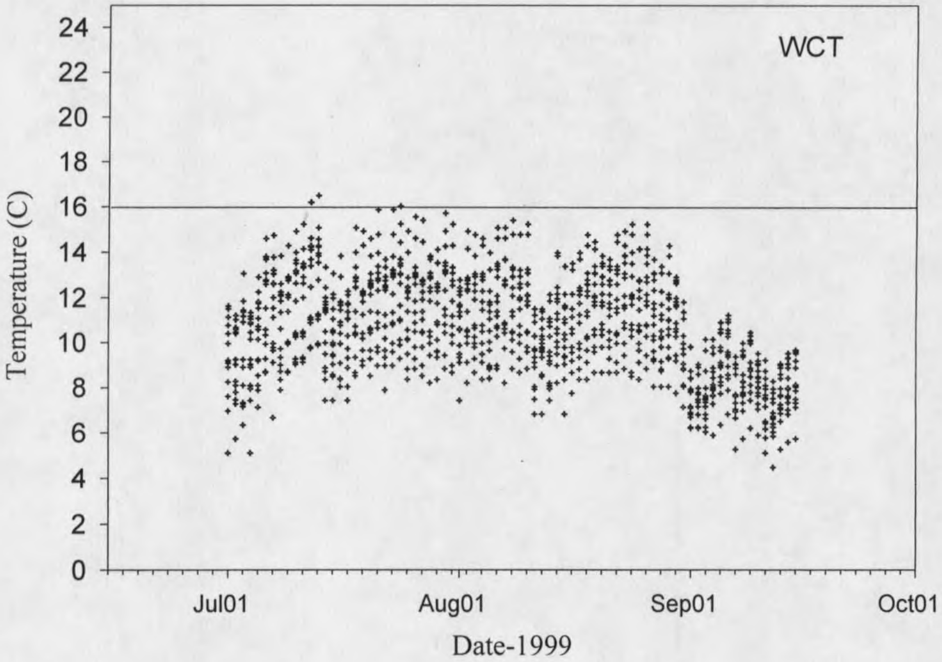


Figure 7. Average daily stream temperatures at sites occupied by westslope cutthroat trout (WCT; n=16) and nonnative salmonids (Nonnatives; n= 25), including a reference line drawn at 12°C.



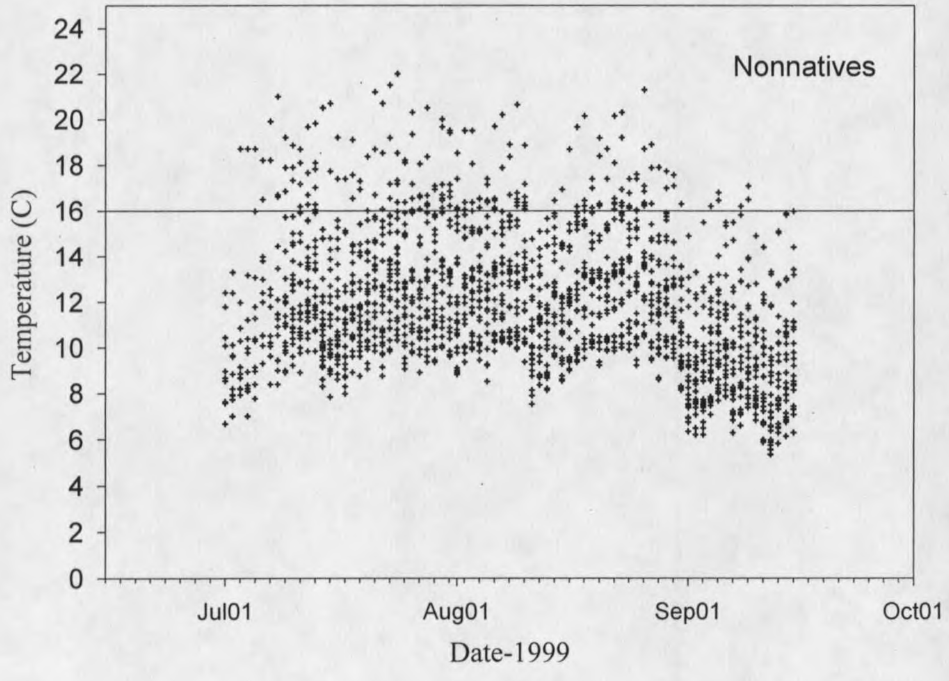


Figure 8. Maximum daily stream temperatures at sites occupied by westslope cutthroat trout (WCT; n=16) and nonnative salmonids (Nonnatives; n= 25), including a reference line drawn at 16°C

When sites occupied by rainbow trout were compared to sites with westslope cutthroat trout all temperature metrics remained significantly lower for sites with cutthroat trout (t-tests, $P<0.05$). The distribution of rainbow trout coincided with a 1-3°C warmer range of stream temperatures than occupied by westslope cutthroat trout. Rainbow trout occupied sites with maximum average daily stream temperatures between 9.24 and 13.1°C, and maximum daily stream temperatures between 12.3 and 18.4°C.

At the basin level, no statistical differences were found between sites occupied by rainbow x cutthroat trout hybrids for any of the temperature metrics examined (t-tests, $P>0.05$). However, in at least one stream, temperature differences corresponded with distribution boundaries of westslope cutthroat trout and nonnative species, including rainbow x cutthroat trout hybrids (Figure 9). Westslope cutthroat trout segregated from nonnative salmonids without the influence of a dispersal barrier in Papoose Creek (Appendix A). In Papoose Creek, thermographs were placed at the upper distribution boundary of westslope cutthroat trout, the upper distribution boundary of nonnative trout species, and at the stream's mouth. Average daily stream temperatures were significantly different at all three sites (ANOVA, $P< 0.001$), with average daily stream temperatures becoming progressively colder at upper stream sample sites. Average daily stream temperatures at the uppermost site in Papoose Creek were also significantly lower than the "coldest" site where nonnative salmonids were captured (Horse Creek, km 8.8) in the Madison River drainage during this study (t-test, $P< 0.001$).

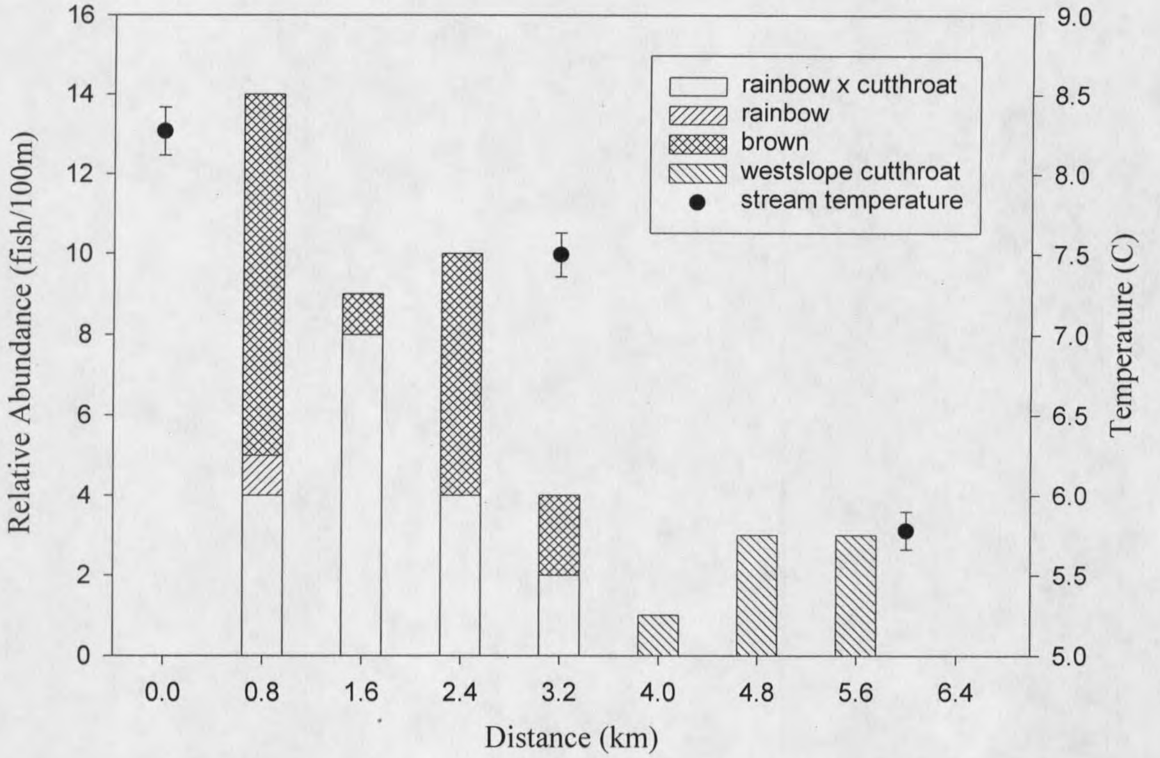


Figure 9. Relative abundance of rainbow, brown, rainbow x cutthroat hybrid, and westslope cutthroat trout greater than 75 mm (left axis) and average summer stream temperatures (right axis) in Papoose Creek by stream kilometer. Vertical lines represent standard errors.

Isolated populations of westslope cutthroat trout encountered a greater range of average summer stream temperatures and slightly warmer stream temperatures (range= 6.6-11.8°C, n=14) than in streams without dispersal barriers (range= 5.9-8.4°C, n=2). These slightly warmer thermal regimes translated into a higher number of degree days at sites above dispersal barriers (mean= 694, SE= 39) than non-isolated sites (mean= 564, SE= 98), but this difference was not statistically significant (t-test, *P*>0.05).

Warm stream temperatures appeared to limit the lower distribution of westslope cutthroat trout in one sub-drainage. In English George and South Fork English George creeks, allopatric westslope cutthroat trout located above a fish dispersal barrier were absent or rare (<1 fish per 100 m of stream) at sites where average daily stream temperatures warmed to 16.0°C, and maximum daily stream temperatures warmed to 24°C during the 1999 sampling season. In contrast, westslope cutthroat trout were moderately abundant (mean abundance= 9 fish per 100 m of stream) in upstream sites where average daily water temperatures remained between 4 and 10°C and maximum recorded stream temperatures remained below 12°C during the summer sampling period.

Stream Temperature Modeling

Water temperature patterns in streams in the Madison River drainage varied considerably both among and within streams (Figure 10). Trends in daily water temperatures at sites measured in multiple years were similar across years. During 1999, stream temperatures reached their annual maximum in July or August, generally corresponding with peaks in air temperature. However, the date on which sites reached their maximum annual stream temperature was well distributed across the summer sampling period (Figure 11). During 1998, sites at which temperatures peaked in late August, 1999, peaked during the first week of September. The two temperature sampling sites monitored in 1997 reached their annual maximum during mid July. Mean stream temperatures were strongly correlated with daily maxima and minima (Table 4). Stream temperatures fluctuated from as little as 2.3 to as great as 16.7°C daily. Ranges of daily stream temperatures were weakly correlated with daily means but were more closely correlated with daily maxima (Table 4). Average summer stream temperatures were

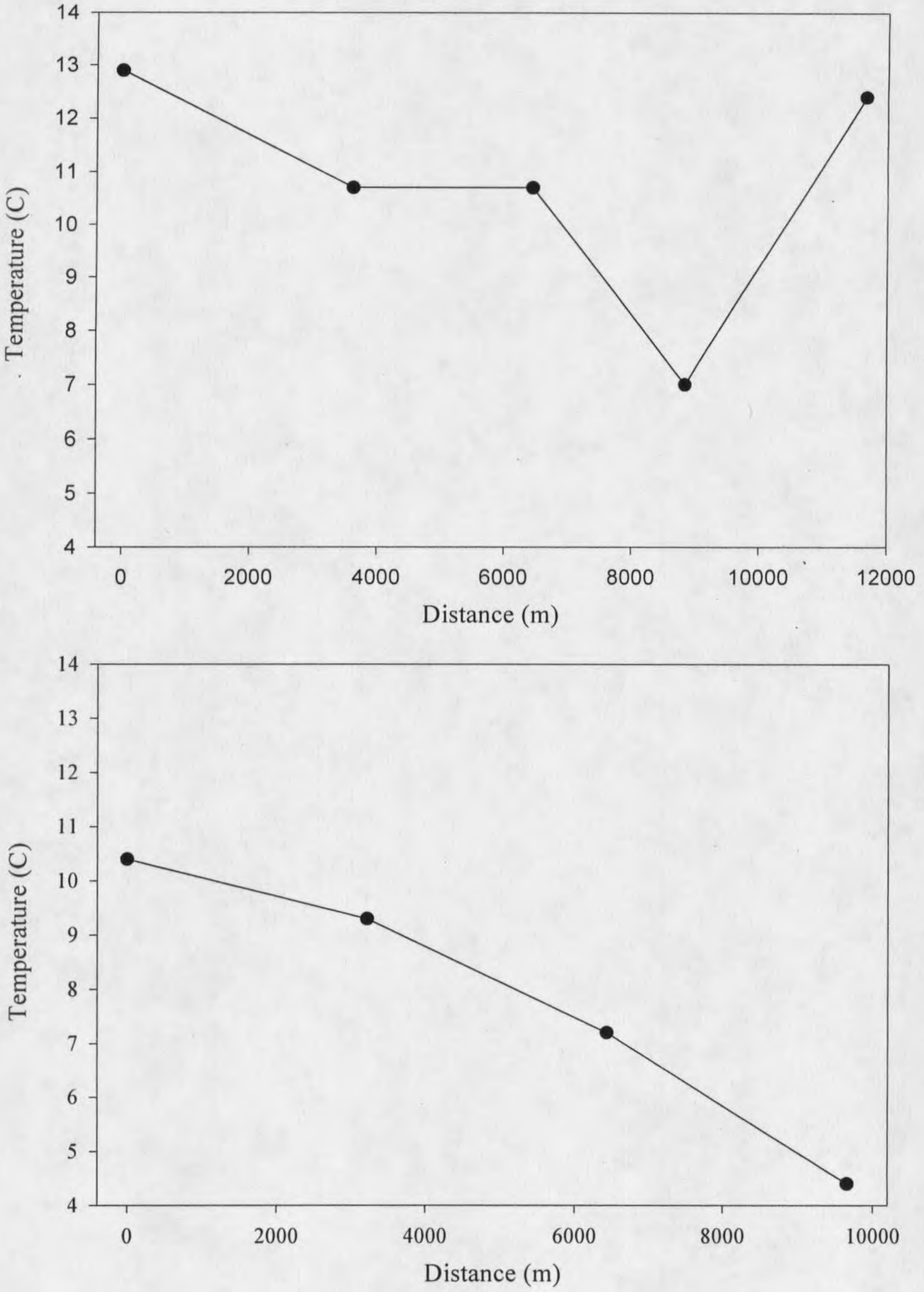


Figure 10. Longitudinal profiles of maximum average daily stream temperatures recorded at five sites in Moose Creek (top) (8 July- 15 September 1999) and four sites in Papoose Creek (bottom) (1 July- 15 September 1999). A headwater lake influenced temperatures in Moose Creek. Distance is measured from the mouth of each stream.

Table 4. Pearson correlations among stream temperature metrics measured at 79 sites on first to fourth order tributaries of the Madison River. Probability values of correlations are given in parentheses.

Metric	Mean	Minima	Maxima	Range
Mean	1.00 -			
Minima	0.90 (<0.01)	1.00 -		
Maxima	0.91 (<0.01)	0.66 (<0.01)	1.00 -	
Range	0.43 (<0.01)	0.01 (0.35)	0.75 (<0.01)	1.00 -

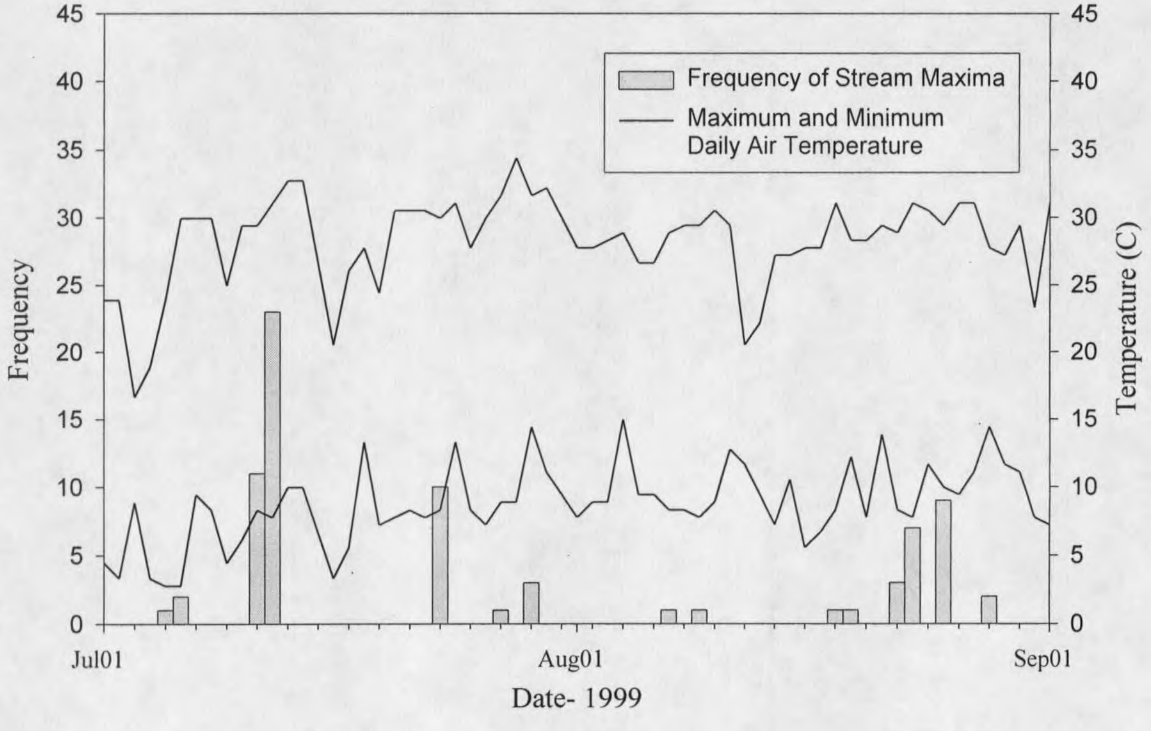


Figure 11. Distribution of annual maximum stream temperatures among temperature recording sites in the Madison River drainage, July 1-September 1, 1999. Corresponding maximum and minimum air temperatures from the NOAA weather station at Ennis, MT are also presented.

significantly colder in streams draining the Madison Range (mean= 7.65°C, SE=0.04) than in the Gravelly Range (mean= 8.63°C, SE=0.05) (t-test, $P<0.001$).

Average daily water temperatures measured at 79 sites (July 1 to September 30) were regressed against a combination of landscape and climate variables (Table 5). These regressions indicated that average daily water temperature could be reliably estimated for streams in the Madison River basin (Table 6). Preliminary analysis identified three sites with unusually warm stream temperatures. These obvious outliers were located near (≤ 1 km) lake outlets and were initially held out of the data set to screen predictor variables. The model selected included average daily air temperature, mean channel elevation, mean channel slope, riparian tree cover, and drainage area. Other model combinations were significant but had lower R^2 or higher AIC values (Table 6). Average daily stream temperatures in the Madison River drainage were related to variables in the final regression as follows (adjusted R^2= 0.76, $P<0.001$):

$$WT = 25.3 + 0.331\,(Air) - 0.00811\,(Elev) - 0.218\,(Slope) - 1.58\,(\%Tree) + 0.00305\,(Area),$$

where, WT= average daily stream temperature (°C), Air= average daily air temperature (°C), Elev= mean channel elevation in meters for the entire stream network above site, Slope= mean channel slope in percent for the entire stream network above site, %Tree= percentage of the riparian tree cover along the entire stream network above site, and DrainageArea= area of watershed above a temperature recording site in square kilometers. All variables included in the final regression were statistically significant (α=0.05) and in a direction (positive or negative) that made physical, as well as, statistical sense. Additionally, multicollinearity resulting from correlations of predictor variables was low (Table 7).

Table 5. Summary of descriptive statistics associated with variables used to model mean daily stream temperature.

Variable	Mean	Standard deviation	Minimum	Maximum
Mean stream temperature (°C)	8.3	2.4	2.3	16.6
Mean air temperature (°C)	17.2	3.8	6.9	23.0
Mean elevation (m)	2411	183.0	1918	2827
Riparian tree (%)	69.7	28.0	0	100
Channel slope (%)	10.7	2.9	6.6	21.8
Drainage area (km)	22.3	25.2	1.3	128.8

Table 6. Stepwise addition of variables used to model average daily stream temperatures in first to fourth order tributaries of the Madison River and corresponding adjusted R^2 and AIC values.

MODEL	R^2 (adjusted)	AIC
WT = Elev	37.13	31496.0
WT = Elev + Air	63.08	24883.0
WT = Elev + Air + Slope	74.65	20210.6
WT = Elev + Air + Slope + %Tree	75.74	19660.9
WT = Elev + Air + Slope + %Tree + Area	75.84	19611.2

Table 7. Pearson correlations among variables used to model average daily stream temperatures in first to fourth order tributaries of the Madison River. Probability values of correlations are given in parentheses.

Variable	Elevation	Slope	%Tree	Drainage area
Elevation	1.00			
	-			
Slope	-0.17	1.00		
	(-0.13)	-		
%Tree	0.08	0.5	1.00	
	(-0.50)	(<0.01)	-	
Drainage area	-0.046	-0.12	0.21	1.00
	(0.69)	(-0.29)	(-0.07)	-

Average daily air temperature accounted for 24% of the variance in the average daily stream temperatures. Maximum and minimum daily air temperatures were omitted as predictor variables because they introduced multicollinearity when used with mean air temperatures and did not produce a better fit when used alone. Air temperature records from the NOAA weather station located at Ennis, MT provided the most reliable estimates of average daily stream temperatures. Although a weather station located at Hebgen Reservoir was closer to thermograph sites located toward the southern end of the Madison Valley, missing records for the sample period precluded its use. Consequently, only air temperature data from the Ennis weather station were used in the final regression. However, when air temperatures from 19 different locations were systematically substituted into the data set, there was a strong linear relationship between the elevation of the air temperature recording station and the intercept term in the resulting regression (intercept= 22.53 + 0.0019*Elevation; $R^2 = 0.83$, $P < 0.001$) indicating that the regression could be corrected by adjusting the intercept term if air temperature from a different locality was substituted into the model.

Channel slope and elevation negatively influenced average daily stream temperatures. The mean elevation of the entire stream network above a stream temperature recording site provided a better model fit than did point elevations of sample sites and was used in the final model. Mean channel elevations at sample sites ranged from 1,928 m to 2,827 m (Table 5). Mean channel elevation made large contributions to reducing the residual and, by itself, accounted for 37.13% of the variation in average daily stream temperatures (Table 6). Mean channel slopes were calculated for the entire stream network above a thermograph site and ranged from 6.6 to 21.8% (Table 5). Including the mean channel slope dramatically improved model fit as measured by both adjusted R^2 and AIC values (Table 6).

Riparian forest cover negatively influenced average daily stream temperatures. Riparian forest cover at sample sites varied from none to 100% (Table 5). Forest cover was measured in 100, 500, and 1000 m reaches located directly above each temperature recording site, as well as along the entire stream network above that site. Model R^2 values increased with reach length, and were maximized when riparian forest cover was calculated for the entire stream network. Therefore this metric was included in the final stream temperature model.

Drainage area had a positive influence on average daily stream temperatures. Correlations between drainage area and stream discharge ($r = 0.89$, $P < 0.001$), and drainage area and stream width ($r = .77$, $P < 0.001$) indicated that drainage area was a reasonable surrogate measure for stream size. Including this variable in the model resulted in small increases in R^2 value, but large decreases in AIC values (Table 6).

Comparison of the model residuals revealed close concordance between estimated and observed average daily stream temperatures. The maximum raw residual value was 4.14°C. However, most estimates closely corresponded with observed stream temperatures. The residual standard deviation (expressed as the root mean squared discrepancy between estimated and observed stream temperatures) was 1.17°C. Since the residuals approximated a normal distribution, this indicates that 95% of the 6,261 average daily stream temperature estimates were within 2.34°C of observed stream temperatures. A plot of the predicted average daily temperatures against observed average daily stream temperature revealed relatively few outliers (Figure 12).

To evaluate the predictive performance of the stream temperature model, I used a jackknife procedure which iteratively removed all observations from a sample site from the data set and estimated stream temperatures at that site using coefficients refit by the remaining data. The standard deviation of the overall jackknife prediction error remained low at 1.26°C. As expected, the Durbin-Watson test for autocorrelation of error terms was positive, confirming that the time series nature of the data prevented independence of

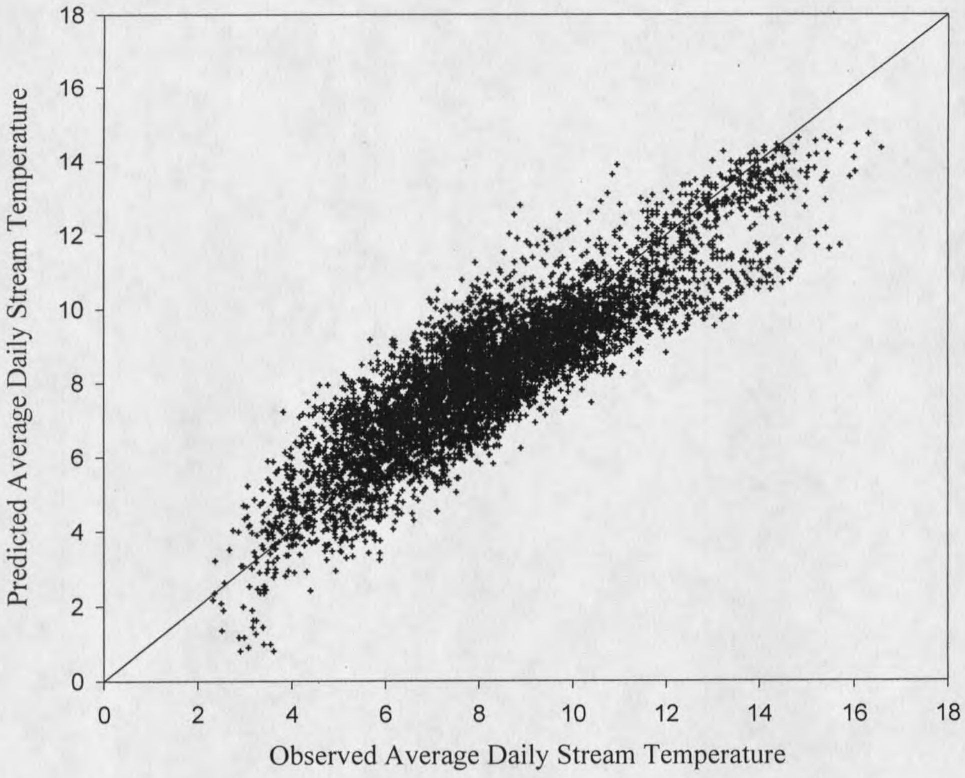


Figure 12. Comparison between observed versus predicted average daily stream temperatures from a multiple regression model using air temperature, elevation, slope, riparian tree abundance, drainage area, and aspect. A 1:1 line is included for reference.

stream temperature observations. However, cross-correlation results from the jackknife procedure indicated that coefficients were very stable and that the model was robust.

When the effect of sample year and sub-drainage were included as random effects in a mixed regression model, all variables considered in the best linear model retained their significance (Table 8). Sample year was not significantly associated with average daily stream temperatures (Table 8). However, the effects of sub-drainage were significantly associated with average daily stream temperatures ($P = 0.004$).

Table 8. Results of a multiple linear regression model ("linear") and a mixed regression model ("mixed" model) to predict average daily stream temperatures. The linear model included elevation, average daily air temperature, channel slope, the percent forest cover, and drainage area. The mixed model included all variables in the linear model plus the "random" variables, year and sub-drainage. The Type III F value and associated probability value are shown for fixed effects and the component of variance, and Wald test for the Z-value and associated probability are shown for random effects. There were a total of 6,210 observations within 16 sub-drainages over 3 years.

Model	Variable	Fixed effects		Random effects		
		Type III F	P	Component of variance	Z-value	P
Linear						
	Elev	9824.02	<0.001			
	Air	6772.97	<0.001			
	Slope	1123.24	<0.001			
	%Tree	302.31	<0.001			
	Drainage area	23.89	<0.001			
	Residual			1.37	55.7	<0.001
Mixed						
	Elev	3666.52	<0.001			
	Air	8066.85	<0.001			
	Slope	482.18	<0.001			
	%Tree	65.72	<0.001			
	Drainage area	26.43	<0.001			
	Year			0.034	0.93	0.172
	Sub-drainage			0.529	2.67	0.004
	Residual			1.16	55.62	<0.001

For sites located below lakes, I assumed that both lake size and the distance below a lake would determine the influence of a lake on stream temperatures. By introducing an interaction between lake area and distance to the lake, I was able to reincorporate sites in close proximity to lake outlets into the dataset. For streams in the Madison River drainage it appeared that lake effects were indistinguishable at locations further than 1.0 km downstream from the lake, and a negative power function approximated this relationship well. The resulting regression was (adjusted $R^2 = 0.76$):

$$WT = 24.3 + 0.332\,(Air) - 0.00783\,(Elev) - 0.222\,(Slope) - 1.49\,(\%Tree) + 0.00405\,(Area) + 3.75\,(LakeArea^{(-0.03084\,LakeDist)}),$$

where, LakeArea= area of standing water connected to the stream channel in square meters, and LakeDist= distance to standing water in kilometers. Parameterization of "LakeDist" involved a type of trial-and-error adjustment to maximize model fit and was first approximated by regressing average daily stream temperatures for the subset of sites below lakes against their distance from the lake.

Comparison of the "lake" model residuals also revealed close concordance between estimated and observed average daily stream temperatures. The residual standard deviation in average daily stream temperature predictions was 1.20°C, indicating that the model was calibrated well to the three sites in close proximity to lake outlets. When sample sites were iteratively removed from the data set and estimated using coefficients refit by the remaining data, the standard deviation of the overall jackknifed prediction error remained relatively low (1.34°C) when the model incorporated "lake" sites. However, cross correlation results from the jackknife procedure indicated that coefficients for the lake size-lake distance interaction were relatively unstable. Jackknifed prediction errors remained relatively high for sites in close proximity to lakes, indicating that predictions for these sites were relatively unreliable.

DISCUSSION

Fish Distribution

The distribution of westslope cutthroat trout in the Madison River drainage between Hebgen and Ennis reservoirs was concentrated in streams draining the Gravelly Range (Figure 5) and was primarily shaped by natural fish dispersal barriers which excluded nonnative salmonids from upstream reaches. I hypothesized that barriers might isolate westslope cutthroat trout from potential hybridization or competition with nonnative salmonids. This appeared to be the case in streams draining the Gravelly Range where the majority of perennial streams supported isolated populations of westslope cutthroat trout. However, except in Cabin Creek where a geologic barrier was located relatively close to the stream's mouth, westslope cutthroat trout did not occur above natural dispersal barriers in the Madison Range. Patterns of fish occurrence indicated that the location of dispersal barriers within a stream network was important in determining the presence or absence of westslope cutthroat trout. While dispersal barriers were equally common in the Gravelly Range (8 of 23 streams) and the Madison Range (13 of 35 streams), fish barriers occurred significantly closer to stream mouths and at lower elevations in the Gravelly Range, which may provide insight into westslope cutthroat trout distribution patterns in the Madison River drainage.

Isolated populations of salmonids face a variety of extinction risks through environmental and demographic variation due to limited physical space and small population sizes associated with fragmented habitats (Rieman et al. 1993). Smaller, more isolated populations are less likely to persist because 1) small populations face a higher risk of extinction through demographic and environmental stochasticity, and 2) isolated populations have no possibility of demographic support or recolonization through dispersal from surrounding populations (Rieman and McIntyre 1995; Dunham et al. 1996). Flood flows, debris torrents, drought, and fires can locally extirpate trout populations (Propst et al. 1992). If westslope cutthroat trout naturally occurred above barriers in streams draining the Madison Range, catastrophic events may have limited cutthroat trout persistence in these areas. However, for many streams it is not known whether westslope cutthroat trout ever had access to reaches above dispersal barriers. In Cherry Creek, a large isolated sub-drainage outside my study area but within the Madison River drainage, native fish are absent from all of the 90 km of contiguous stream habitat above an 8 m high barrier (Bramblett 1998). Because of the large size and hydrologic complexity of this sub-drainage (Bramblett 1998), the absence of native fish species above this barrier strongly suggests that westslope cutthroat trout were historically absent above the geologic falls rather than extirpated due to stochastic events. In my study, fishless reaches above dramatic geologic waterfalls exceeding 10 m high in several streams within the Indian Creek sub-drainage (Figure 5) may also represent sites which were never colonized by westslope cutthroat trout. Consequently, it is unlikely that all fishless reaches in the Madison Range have resulted from localized population extinctions.

Nevertheless, local extirpations of isolated salmonid populations as a result of catastrophic events have been documented elsewhere (e.g., Propst et al. 1992). Kruse et al. (1997) found that wild Yellowstone cutthroat trout were absent above natural dispersal barriers in the Wood and Greybull river drainages, Wyoming. They attributed this absence to relatively short stream lengths above barriers, poor habitat conditions, and relatively common occurrences of catastrophic events. Similarly, Dunham et al. (1996) suggested that the general absence of Lahontan cutthroat trout *O. c. henshawi* populations above natural dispersal barriers was likely a byproduct of high extinction and low recolonization or population rescue probabilities in such small, isolated habitats. In my study, despite apparently suitable physical habitat (Sloat et al. 2000) fish were absent above a relatively recent barrier formed by a large debris jam in Wolf Creek (Figure 5), suggesting that westslope cutthroat trout had been eliminated from this historically accessible stream reach.

While not all fishless stream reaches above barriers represent sites where cutthroat trout have been extirpated, this does not diminish the risk of extinction for small, geographically restricted populations. Where westslope cutthroat trout currently exist above barriers in the Madison River drainage, low population sizes and isolation may place many of these populations at risk. Although abundances of westslope cutthroat trout were not significantly lower than those of nonnative salmonids, relative abundances of all salmonid species in the Madison River drainage were generally much lower than in tributaries from other drainages in the upper Missouri and upper Clark Fork river basins in Montana (Sloat et al. 2000). The low abundance of trout in Madison River tributaries may be related to the relatively high elevation of this river basin, inherent geologic instability that translates into somewhat unstable stream channels, and moderate to low productivity of its watersheds (Sloat et al. 2000).

In most streams, westslope cutthroat populations existed in relatively short stream reaches (mean occupied length= 4.51 km). Based on an empirical evaluation of translocation success, Harig and Fausch (2001) suggested that stream segments under 5.7 km long may have insufficient space to sustain adult and juvenile greenback cutthroat trout *O. c. stomias*. Hilderbrand and Kershner (2000) developed a simple relationship between observed cutthroat trout abundances, the proportion of individuals leaving a population through emigration and mortality, and desired population sizes to estimate the minimum stream length (MSL) necessary to maintain viable cutthroat trout populations. They recommended a population benchmark of 2,500 individuals >75 mm long to insure the long term persistence of isolated populations. Based on a target population size of 2,500 individuals, and assuming no proportional loss of individuals, only two streams sampled in this study have MSL's that meet criteria for long term persistence presented by Hilderbrand and Kershner (2000) (Table 9).

Insufficient space does not necessarily mean that a population will become extinct (Hilderbrand and Kershner 2000). Some fish populations have persisted for extended periods in small habitat patches isolated by natural barriers and may have adapted to restricted space (Northcote et al. 1970; Northcote 1981, 1992). Northcote (1981) reported that heritable differences in rheotaxis between rainbow trout populations above and below a waterfall were genetically coded. Salmonid populations introduced into barren lakes may develop both inlet- and outlet-spawning populations with heritable

Table 9. Mean fish abundance (>75 mm total length) per linear meter of stream used for the minimum stream length estimator (MSL), and observed occupied stream lengths (including inhabited tributaries) for westslope cutthroat trout populations sampled in this study. Bold streams meet the MSL recommended by Hilderbrand and Kershner (2000).

Stream	Mean abundance[1] fish/m	Occupied length km	MSL km
Arasta Creek	<0.10*	2.4	>25.0
Cabin Creek	**0.31**	**20.0**	**8.1**
English George Creek	0.25	9.0	10.0
Horse Creek	0.20	7.1	12.5
Hyde Creek	0.36	2.7	6.9
Papoose Creek	0.10	2.4	25.0
South Fork Indian Creek	<0.10*	2.4	>25.0
Soap Creek Creek	0.21	3.4	11.9
Standard Creek	**0.28**	**12.1**	**8.9**
Trail Fork Bear Creek	<0.10*	<1.0	>25.0
Wall Creek	0.20	4.8	12.5

[1] From multiple pass depletion estimators except * where no estimates were made because very few fish were captured.

differences in the direction that fry move to reach the lake (Kaya 1989). Similarly, Shepard et al. (1998) found that the proportion of stream dwelling westslope cutthroat trout moving 0.5 km or longer was negatively correlated to the level of isolation experienced by the population. While these local adaptations may be advantageous for individuals in restricted habitats, adaptations to stochastic events such as extreme floods, debris flows, or droughts may be unlikely because either the intensity or the time between such events is too great (Poff 1992). Additionally, traits which confer the greatest advantages to species occupying marginal habitats, such as high mobility and multiple life histories (Thorpe 1994), may actually be selected against in isolated habitats. Consequently, without the chance for recolonization, population extinctions in fragmented stream systems may proceed in a “ratchet-like” manner, increasing the chances of basin-wide extinction (Dunham et al. 1996).

Translocations of trout into fishless reaches above natural barriers are a common management action to increase the range of native fishes (Harig 2000). The general absence of fish from high elevation reaches above fish barriers found in this and other studies of cutthroat trout (Dunham 1997; Kruse et al. 1997) indicate that this action may not guarantee the long-term viability of cutthroat trout populations (e.g., Harig 2000). However, results from my study also indicate that dispersal barriers may effectively protect westslope cutthroat trout populations when located relatively low within stream networks. While isolation carries risks associated with low population sizes and limited physical space, it is often the only factor preventing displacement by nonnative salmonids through competition and hybridization. For example, Hanzel (1959) found that the majority of pure cutthroat trout populations in Montana occurred above fish dispersal barriers. Young et al. (1996) reported that 20 of 27 allopatric populations of genetically pure Colorado River cutthroat trout *O. c. pleuriticus* considered indigenous, and in a drainage not recently stocked, were located above fish migration barriers. Distribution patterns I observed in the Gravelly Range illustrate the importance of natural barriers to remaining westslope cutthroat trout populations in the Madison River drainage. Primarily because of their association with barriers occurring relatively low within stream networks, isolated populations of westslope cutthroat trout occupied greater stream lengths and reached significantly higher abundances than non-isolated cutthroat trout populations. However, based on minimum habitat requirements suggested by other researchers (e.g., Harig 2000; Hilderbrand and Kershner 2000) the viability of most westslope cutthroat trout populations in the Madison River drainage remains tenuous and, where possible, these populations should either be expanded further downstream or replicated in larger drainages provided that potential hybridizing and competing species are first removed.

Unfortunately, in Madison River tributaries even some populations isolated by dispersal barriers were slightly introgressed (Appendix A), indicating that nonnative trout have been widely introduced into headwater habitats throughout the drainage. Consequently, additional sampling is necessary to determine if upper Arasta and Standard creeks still support pure populations of westslope cutthroat trout. The genetic status of many other westslope cutthroat trout populations in the Madison River drainage remains somewhat uncertain due to the possibility that some of these populations may contain a "deviant allele" that is a diagnostic allele characteristic of rainbow or Yellowstone cutthroat trout but that may simply be a rare westslope cutthroat trout genetic variation (Appendix A). This situation likely exists for populations in upper English George, Papoose and Wall creeks, and may exist for a few other populations (Appendix A). Additional genetic sampling will be necessary for some of these populations to clarify their genetic status before population expansion or replication efforts are undertaken.

Stream Temperature

In addition to dispersal barriers, stream temperature also influenced westslope cutthroat trout distribution in the Madison River drainage. The association of most westslope cutthroat trout populations with fish dispersal barriers in the Madison River drainage obviously confounds my ability to make direct temperature or species interaction inferences. Overall temperature relationships would become clearer and stronger in areas where westslope cutthroat trout and other salmonids segregated without the influence of barriers. However, in at least one stream, longitudinal temperature patterns coincided with fish distribution boundaries. Genetic data from Papoose Creek suggest that pure westslope cutthroat trout segregated from nonnative salmonids without the presence of a fish dispersal barrier (Appendix A). Stream temperatures in Papoose Creek were significantly lower in the reach occupied by westslope cutthroat trout than in downstream reaches occupied by nonnative salmonids. Stream temperatures in the reach of Papoose Creek occupied by westslope cutthroat trout were also significantly lower than the coldest site occupied by nonnative salmonids in the Madison River drainage.

This situation may also occur in South Fork of Indian Creek (see Figure 5), where genetically pure westslope cutthroat trout were observed upstream from reaches occupied by nonnative species. However, the remote locality and lack of *a priori* knowledge of fish distribution and genetic status precluded temperature measurement in this stream. Similar to my findings, Mullan et al. (1992) found that in naturally sympatric populations, rainbow trout excluded the first two or three age classes of westslope cutthroat trout up to a point where stream temperatures decline to about 1,600 annual thermal units (sum of average daily temperatures (°C)).

These distribution boundaries may be attributable to temperature mediated competitive differences between cutthroat trout and nonnative salmonids or temperature mediated growth differences. For example, Destaso and Rahel (1994) found that a 1°C difference in Critical Thermal Maxima (CTM) between brook trout *Salvelinus fontinalis* and cutthroat trout correlated with greater competitive ability of brook trout at warmer temperatures. Adams (1999) found that lower growth and fecundity, and greater female age-at-maturity resulting from cold stream temperatures limited upstream invasions of brook trout in some Rocky Mountain streams. In addition to brook trout, thermal tolerances of cutthroat trout are generally lower than nonnative species such as rainbow, and brown trout (Feldmuth and Erikson 1978; Eaton et al. 1995).

Although there was considerable overlap, and despite the confounding influence of dispersal barriers, all stream temperature metrics tested were significantly lower at sites occupied by westslope cutthroat trout than at sites occupied by nonnative salmonids (Table 3). When sites occupied by rainbow trout, a potential hybridizing and competing species, were compared to sites with westslope cutthroat trout all temperature metrics remained significantly lower for sites with cutthroat trout. The distribution of rainbow trout coincided with a 1-3°C warmer range of stream temperatures than occupied by westslope cutthroat trout. Magnuson et al. (1978) considered the "fundamental thermal niche" for fishes to encompass 4°C, and Christie and Regier (1988) suggested this niche ranged from –3 and +1°C around a species optimal growth temperature. For rainbow trout, maximum growth occurs at approximately 17.2 °C (Hokanson et al. 1977), and thus the range of maximum daily temperatures occupied by rainbow trout in the Madison River drainage (12.3-18.4°C) corresponded closely with their fundamental thermal niche (14.2-18.2°C). Consequently, competitive advantages of rainbow trout at higher temperatures near their optimal growth range may account for the absence of westslope cutthroat trout where rainbow trout were found.

Contrary to patterns for rainbow trout, no statistical differences were found between sites occupied by rainbow x cutthroat trout hybrids and those occupied by westslope cutthroat trout at the basin level. The influence of genetic introgression of both rainbow trout and Yellowstone cutthroat trout on the thermal response of westslope cutthroat trout has not been studied. In laboratory experiments, Ihssen (1973) found that two reciprocal first generation hybrids of brook trout and lake trout *Salvenlinus namaycush* had similar times to death upon exposure to several lethal high temperatures for a series of acclimation temperatures. Second generation hybrids were intermediate to the parent species in resistance and the backcrossed offspring were intermediate between the second generation hybrids and their respective parents. This suggests that differences in thermal responses between potentially hybridizing species may quickly break down when hybrid swarms develop. If these patterns are similar for rainbow x cutthroat trout hybrids, there is a need to differentiate relatively pure from hybridized populations when investigating relationships between cutthroat trout distribution and stream temperature. Some populations with relatively high (but less than 90%; see Appendix A) proportions of westslope cutthroat trout genetic material were classified as nonnative salmonids, which may have weakened relationships between stream temperatures and fish distribution.

Basin-wide, westslope cutthroat trout were associated with habitats where average daily stream temperatures generally remained below 12°C and maximum daily stream temperatures remained below 16°C. Bell (1984) reported a preferred temperature range of 9 to 12°C for cutthroat trout. Dwyer and Kramer (1975) reported the greatest scope for activity in cutthroat trout occurred at 15°C when tested at 5, 10, 15, 20, and 24°C. Assuming that the scope for activity was a better measure of optimal temperature than temperature preference tests, Hickman and Raleigh (1982) selected 12 to 15°C as an optimal temperature range for cutthroat trout. Average and maximum daily water temperatures at sites occupied by westslope cutthroat trout generally corresponded with these reported ranges of preferred and optimal temperatures for cutthroat trout.

While westslope cutthroat trout were associated with habitats where stream temperatures seldom exceeded 16°C, this should not be construed as the upper thermal tolerance limit for this subspecies. Although warm stream temperatures approaching 25°C appeared to limit the downstream distribution of westslope cutthroat trout in the

English George sub-drainage, temperatures in most reaches now occupied by nonnative species were well below reported critical thermal maxima of 27-28°C for cutthroat trout (Feldmuth and Eriksen 1978; DeStaso and Rahel 1994). The patterns of fish occurrence and stream temperature I observed indicate that westslope cutthroat trout have been displaced from warmer stream habitats and that westslope cutthroat trout now occupy a narrower and colder range of stream temperatures than they did historically. In my study, isolated populations of westslope cutthroat trout encountered a higher and greater range of average summer stream temperatures than in streams without dispersal barriers, indicating that without the influence fish barriers the range of stream temperatures occupied by westslope cutthroat trout would be substantially narrower and colder yet due to the influence of nonnative salmonids.

While many researchers have focused on the role maximum stream temperatures play in regulating salmonid distribution (e.g., Dunham et al. 1999; Haas, in press), few have explicitly addressed the ecological costs for salmonids in habitats where stream temperatures remain below thermal optima. Several westslope cutthroat trout populations sampled in this study inhabited streams where water temperatures remained below optimal temperature ranges (Hickman and Raleigh 1982) for most of the summer season (Figures 7 and 8). Low westslope cutthroat trout densities in Papoose Creek and Trail Fork of Bear Creek (Appendix B) may be attributed to low stream temperatures, since maximum stream temperatures remained below 10°C throughout the summer at sites where westslope cutthroat trout were captured in these two streams.

The two major external factors controlling fish growth are water temperature and food availability (Weatherly and Rogers 1978). Averett (1963) documented higher growth rates for westslope cutthroat trout from lower versus higher elevation tributaries of the St. Joe River, Idaho, presumably a result of differences in stream temperatures. Body size is strongly related to fecundity in westslope cutthroat trout (Downs 1995). Cold stream temperatures can delay cutthroat trout spawning, prolong egg incubation (Behnke 1992; USFWS 1998; Harrig 2000), and reduce embryo survival (Hubert et al. 1994; Stonecypher et al. 1994). Late hatching fry risk winter starvation if they cannot grow enough to withstand metabolic deficits at low winter temperatures (Cunjak and Power 1987; Shuter and Post 1990; Harrig 2000). Consequently, westslope cutthroat trout probably experience lower individual fitness and reproductive success in habitats where temperatures remain well below optimal ranges. The low abundances of westslope cutthroat trout I observed at sites not physically isolated from nonnative species suggest that, while colder stream temperatures may provide a competitive or demographic boost for westslope cutthroat trout relative to nonnative species, sub-optimal thermal regimes may also limit a population's ability to buffer environmental and demographic stochasticity in headwater habitats.

In addition to fish dispersal barriers, other local factors may affect the correspondence between fish distributions and temperature within streams, including variability of habitat quality, disease, food availability, and water quality and quantity (Dunham 1999). The potential for seasonal migrations may also add noise to data relating fish distributions directly to stream thermal characteristics (Dunham 1999).

Northcote (1992) noted that the most extensive movements in resident salmonid populations were associated with spawning migrations. However, Downs (1995) reported that westslope cutthroat trout living in headwater habitats did not appear to have extended spawning migrations. Similarly, Shepard et al. (1998) found that while some individual westslope cutthroat trout move relatively long distances, little movement was observed for most resident westslope cutthroat trout inhabiting headwater streams in Montana.

A potential problem with my study is a lack of temporal concordance between fish distribution and temperature data. I matched fish sampling records with temperature records corresponding most closely in time. Since fish were sampled over a 3 year period, while the majority of the temperature data were collected in 1999, stream temperatures were not measured during the same year as the fish sampling event in some locations. For the temperature associations presented in this study to be valid, two assumptions must be met. First, fish distribution boundaries did not change during the period of my study. Other studies have found that distribution limits of cutthroat trout were relatively constant across a 20-year period (1977-1997) despite fluctuations in densities (Dunham et al. 1999). Similarly, brook and rainbow trout showed no net change in distribution limits over a similar time period in eastern Tennessee streams (Strange and Habera 1998). I expect this to be true in the Madison River drainage as well, especially considering the relatively short time period of my study and the strong influence of dispersal barriers on fish distribution. The second assumption is that measured stream temperatures are representative of temperatures experienced by fish during the year fish distribution data were collected. This assumption also seems reasonable since published air temperature records for the period of my study indicate that annual and summer air temperatures from 1997-1999 corresponded closely with long term average air temperatures (NOAA 1997, 1998, 1999).

Stream Temperature Modeling

Several investigators have developed models to predict site-specific water temperatures across relatively broad geographic regions. Hawkins et al. (1997) used channel reach characteristics to predict July thermal characteristics in 45 montane streams in California. Wehrly et al. (1998) used catchment- and local-scale landscape variables to predict July weekly average maximum and minimum stream temperatures in Michigan's lower peninsula. Isaak and Hubert (2001) used geomorphic characteristics and landscape features to predict stream temperature maxima in montane streams in Wyoming. However, few investigators have developed models that predict site specific daily stream temperatures Stefan and Preud'home (1993) used air-water temperature relationships to predict daily temperatures in 11 streams in the Central United States. My study is the only one I am aware of that predicts site specific daily stream temperatures across broad geographic areas in montane landscapes. The model developed in my study, using GIS derived landscape variables and published climate records, provided reliable estimates of average daily stream temperatures in the Madison River drainage.

Accurate estimates of stream temperature can be obtained using empirical models based on air-water temperature relationships (e.g., Holtby 1988; Stefan and Preud'homme 1993; Journdanais et al. 1998; Hennessey 1998), or heat budget models (Brown 1969; Kothandaraman 1972). However, these models are site specific and, in the case of heat budget model, may require extensive data inputs. Additionally, these models are better suited for predicting temporal changes in temperature at a site rather than spatial variation in temperature across multiple sites (Wehrly et al. 1998). Predicting site-specific temperatures across a broad region (especially in diverse montane landscapes) using heat budget calculations or air-water temperature relationships requires the development and calibration of multiple models (Hennessey 1998; Wehrly et al. 1998). Consequently, these types of models cannot efficiently generate broad-scale temperature coverage.

In my analysis, average daily air temperature drove the temporal variability in average daily stream temperature predictions and had a strong positive effect on stream temperatures. Regression analysis indicated that relationships between air and water temperatures were influenced by the elevation of the air temperature recording station and that this influence was predictable. This indicates that relationships between average daily air and water temperatures I observed in the Madison River drainage can be adjusted if this temperature model is applied to other areas of interest. Montana has over 300 weather stations that maintain daily air temperature records. Coupled with GIS derived landscape variables, such air temperature data can be used to make stream temperature predictions in other areas where stream temperature observations have been made to validate the relationships I observed in this study.

Elevation had a negative effect on stream temperatures and accounted for a large proportion of the spatial variability in average daily stream temperatures. I found that mean channel elevations correlated more strongly with stream temperatures than did point elevations, a result similar to that of Isaak and Hubert (2001), who attributed this stronger correlation to the spatially distributed effect of air temperature on stream temperature. Point elevations are commonly used as surrogates for stream temperatures when actual stream temperature data are not available (c.f., Fausch et al. 1988; Isaak and Hubert 2001). My results agree with Isaak and Hubert's (2001) recommendation that mean elevations become the preferred surrogate whenever stream temperature data are unavailable.

Channel slope also affected average daily stream temperatures. In my analysis, sites having relatively higher channel gradients had cooler stream temperatures, a result similar to Wehrly et al. (1998). Channel slope is correlated with velocity and serves as a proxy for the amount of time that water spends in the channel as it flows downstream (Wehrly et al. 1998). The time it takes water to move downstream is the time a unit volume of water is exposed to heat exchange with the atmosphere (Smith and Lavis 1975; Theurer et al. 1984; Bartholow 1989). In general, as travel time increases, streams accumulate heat and approach an equilibrium temperature at which net heat exchange with the atmosphere is zero (Brown 1969; Theurer et al. 1984; Bartholow 1989). Factors that reduce travel time, such as higher channel gradients, reduce the amount of time a unit volume of water is exposed to sources of heat.

In this analysis, sites having a greater percentage of riparian forest cover had lower average daily stream temperatures. Riparian vegetation intercepts direct solar radiation that would otherwise be absorbed at the stream surface (Wehrly et al. 1998) and therefore streams with greater riparian forest cover should have lower stream temperatures. Several studies have documented the localized effects of tree shading on stream temperatures (Brown and Krygier 1970; Feller 1981), but findings from my study agree with other studies (Wehrly et al. 1998; Isaak and Hubert 2001) suggesting that landscape-level effects of this factor are also important.

Drainage area had a positive effect on average daily stream temperatures and was strongly correlated with both stream discharge and stream width. Increases in stream width may increase stream temperatures since riparian vegetation provides less shade to wider streams, while the volume of water in the stream channel dictates the rate the stream temperatures approach equilibrium with air temperature (Wehrly et al. 1998). However, all other factors being equal, streams with larger drainage areas have longer stream channels and therefore more time to equilibrate with ambient air temperature. Therefore as drainage area increases, stream temperatures should increase. I found that streams with larger drainage areas had higher average daily stream temperatures.

Channel aspect may influence stream temperatures if stream orientation relative to the path of the sun affects the amount of solar radiation absorbed at the stream surface (Johnson 1971; Smith and Lavis 1975). However, stream aspect was not a significant predictor of average daily stream temperatures in my study. Other investigations have also documented a weak effect of stream aspect on water temperature (Johnson 1971; Smith and Lavis 1975; Isaak and Hubert 2001). The effect of channel aspect is probably confounded by riparian vegetation that mediates the amount of solar radiation that reaches a stream's surface (Isaak and Hubert 2001). Additionally, streams in the Madison River drainage have predominately easterly or westerly aspects and are oriented similarly relative to the path of the sun, so differences in stream temperature related to stream aspect may not be apparent.

I did not detect a year effect on average daily stream temperatures. Similarly, Wehrly et al. (1998) found no significant effect of sampling year on stream temperatures when stream temperatures were collected over a 7 year period (1989-1996). Interannual variation in air temperature would be expected to lead to variation in stream temperatures at a site. The fact that daily air temperatures were included in the stream temperature model probably accounts for this lack of a sample year effect.

My study indicates that average channel characteristics, for the entire network upstream from each site, are important factors controlling stream temperatures, and that most of the variation in average daily temperatures of mountain streams can be explained by macroscale factors. However, unexplainable variation from three sources will always exist: 1) unique attributes of particular landscapes, 2) microscale factors, and 3) measurement errors (Isaak and Hubert 2001).

Despite considerable variation within some streams, random effects of drainage were detected in a mixed regression model, indicating that some spatial correlation of temperatures occurred among sites. Positive spatial autocorrelation may not bias parameter estimates, but may underestimate variances, inflating the chance of finding a significant result when in fact one does not exist: a type I statistical error (Zar 1984; Dunham et al. 1997). However, after accounting for the effects of drainage in a mixed regression model, all of the fixed variables included in the linear regression retained their statistical significance. This spatial autocorrelation indicates that unique attributes of each drainage that I did not measure may also influence average daily stream temperatures.

Microscale factors related to beaver dam complexes and local channel form (Hawkins et al. 1997), aquifer recharge from seeps and springs that contribute coldwater inflows over the course of a stream (Bilby 1984), and headwater lakes alter how streams adjust to ambient air temperatures and solar influxes. While I attempted to account for, or avoid confounding factors at this scale, microscale factors important to stream temperatures are sometimes difficult to detect (Isaak and Hubert 2001). However, while results from my study don't negate the importance of microscale factors, the high concordance between observed stream temperatures and estimates from my model indicate that microscale factors do not produce an overriding influence on stream temperatures at the basin-scale.

The final source of unexplained variation is simply due to measurement errors associated with the variables used to predict average daily stream temperatures. In this study, I did not directly measure the factors that control stream temperature. Instead, I used GIS derived landscape variables that were highly correlated with stream temperature and where applicable, assumed that a causal relationship existed between each independent variable and average daily stream temperature. For example, I did not directly measure the extent of shading provided by riparian vegetation. Instead, I assumed that the percentage of tree cover derived from the landcover layer was directly related to the amount of tree cover in the area adjacent to the stream and that this was proportional to the extent of riparian shading. Likewise, I did not use site measurements of stream width or discharge, but instead assumed that drainage area was a reasonable surrogate for these measures of stream size.

Conclusion and Recommendations

This study provided important information on the distribution and abundance of westslope cutthroat trout in the Madison River drainage. Identifying and protecting existing populations is the first step in an effective conservation plan for westslope cutthroat trout (MFWP 1999). Because of the inherent risks associated with the restricted distribution and small sizes of many westslope cutthroat trout populations, simply maintaining the status quo will probably not be sufficient to promote the long-term persistence of all populations. Due to the limited number of genetically pure populations of westslope cutthroat trout in the Madison River drainage, I believe it would be worthwhile to replicate existing pure populations. I recommend that further genetic testing be completed in English George, upper Papoose, upper South Fork Indian, and Wall creek sub-drainages to confirm the presence of genetically pure populations in these areas. Should any of these populations prove to be genetically pure, they should be replicated, preferably somewhere within the Madison River drainage, as soon as technically feasible to conserve these unique genetic resources. When westslope cutthroat trout populations are to be expanded, results from my study agree with others (MFWP 1999; Harrig 2000; Hilderbrand and Kershner 2000) suggesting that translocation sites be located relatively low within stream networks to insure that habitat space and quality are sufficient to maintain the long-term viability of cutthroat trout populations. Additionally, I recommend that existing genetically pure populations of westslope cutthroat trout be expanded downstream, where possible, to incorporate larger habitat areas. I also recommend that slightly introgressed (<10% introgression) westslope cutthroat trout populations be managed with the same protection given to genetically pure westslope cutthroat trout, because such populations may have genetic value and their presence indicates suitable habitat for westslope cutthroat trout (Shepard et al. 1997; MFWP 1999).

Despite some shortcomings, my study provided important information on the thermal regimes associated with suitable habitat for westslope cutthroat trout as well as evidence that distribution boundaries between westslope cutthroat trout and nonnative salmonids are related to stream temperatures. However, relationships between westslope cutthroat trout distribution and abundance and stream temperature need to be clarified through both laboratory experiments and more extensive field studies.

Because temperature data acquisition can be expensive and time consuming, the predictive stream temperature model developed in this study will be a valuable, cost effective tool for resource managers. At present this model has only been tested with data from July through September in the Madison River drainage and caution must be exercised in the generality of its application beyond the summer season and in other areas until replicate tests are conducted. Future testing and refinement should result in better predictive models.

When the thermal requirements of westslope cutthroat trout are better known, the stream temperature model developed in this study can be used to prioritize westslope cutthroat trout conservation efforts at basin-wide scales by: 1) predicting westslope cutthroat trout occurrence in areas where their distributions are unknown; 2) identifying stream reaches where translocations of westslope cutthroat trout have a high probability of success; and 3) predicting effects of land use and global warming on westslope cutthroat trout distribution and abundance.

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APPENDICES

APPENDIX A

GENETIC TESTING RESULTS FOR WESTSLOPE CUTTRHOAT TROUT POPULATIONS IN THE MADISON RIVER DRAINAGE

Appendix A. Genetic testing results for sites in the Madison River drainage by date, location (Legal or stream kilometer), sample size (n), and analysis method (E = allozyme electrophoretic and P = PINE DNA), showing species code (RB = rainbow trout; WCT = westslope cutthroat trout; and YCT = Yellowstone cutthroat trout) and proportion of sample estimated to contain alleles characteristic of each species (NA = proportions not available), and, where applicable, number of individuals that were pure WCT. Information from the Montana Resource Information System database (http://www.nris.mt.us) unless otherwise denoted.

STREAM				Genetic Results (species code and %)			
Date	Location	n	Analysis method	Code %	Code %	Code %	Number pure WCT
ARASTA CR							
7/26/1995	07S03W36[1]	5	E	WCT 100			5
7/20/1999	08S02W06[2]	1	P	WCT *NA*		YCT *NA*	0
BUFFALO CR							
7/26/1995	07S02W31[1]	4	E	WCT 100			4
7/20/1999	08S02W05[2]	7	P	WCT 84	RB 4	YCT 12	0
7/20/1999	07S02W31[2]	7	P	WCT 84	RB 4	YCT 12	0
CABIN CR							
8/31/1997	11S04E05	7	E	WCT 100			7
11/15/1998	11S03E15	8	E	WCT 0	RB 71	YCT 29	0
4/19/1999	11S03E14	10	P	WCT 93	RB 7	YCT 0	0
7/26/1999	Km 3.2-9.3	27	P	WCT 96	RB 4	YCT 0	0
7/27/1999	11S04E14[3]	6	P	WCT>90	RB<10		5
CABIN CR, M FK							
6/01/1993	11S04E11	10	E	WCT 100			10
7/27/1999	Km 0-8.0[3]	58	P	WCT 98	RB 2	YCT 0	0
CORRAL CR							
7/8/1998	Km 9.7[4]	21	P	WCT 86	RB 8	YCT 6	0
ENGLISH GEORGE CR							
8/1/1992	09S01W36	15	E	WCT 95	RB 5	YCT 0	0
6/8/1999	10S01W02[5]	10	P	WCT>90	RB<10		*NA*
HORSE CR							
8/10/1995	10S02W19[1]	8	E	WCT 100	RB 0	YCT 0	8
7/28/1998	Km 7.2-11.3[6]	70	P	WCT 88	RB 3	YCT 9	0
7/28/1998	Km 12.1-13.7[6]	29	P	WCT 98	RB 0	YCT 2	*NA*

Appendix A: (Continued)

STREAM			Analysis	Genetic Results (species code and %)			Number pure
Date	Location	n	method	Code %	Code %	Code %	WCT
HYDE CR							
7/21/1995	09S01W34[2]	3	E	WCT 96	RB 4	YCT 0	0
7/13/1999	09S01W33[3]	16	P	WCT 96	RB 0	YCT 4	0
MIDDLE FORK BEAR CR							
7/27/1994	07S02E06[7]	2	E	WCT 87	RB 13	YCT 0	0
NORTH FORK BEAR CR							
7/26/1994	07S01E36[7]	4	E	WCT 70	RB 25	YCT 5	0
PAPOOSE CR							
7/26/1994	11S02E06[7]	4	E	WCT 100	RB 0	YCT 0	4
7/27/1999	Km 0-5.6[8]	24	P	WCT *NA*	RB *NA*	YCT *NA*	6
QUAKING ASPEN CR							
6/30/1998	Km 1.6	16	P	WCT 77	RB 23	YCT 0	0
SOAP CR							
9/19/1991	11S01E29	12	E	WCT 99	RB 0	YCT 0.9	0
9/01/1992	11S01E29	16	E	WCT 99	RB 0	YCT 0.6	0
SOUTH FORK ENGLISH GEORGE CR							
6/8/1999	10S01W02[9]	9	P	WCT *NA*	RB *NA*	YCT *NA*	*NA*
SOUTH FORK INDIAN CREEK							
8/05/1998	Km 1.6-4.0[10]	22	P	WCT 79	RB 15	YCT 6	0
8/05/1998	Km 4.0-5.6[10]	12	P	WCT >90	RB *NA*	YCT *NA*	*NA*
STANDARD CR							
8/11/1997	11S01E05[11]	13	E	WCT *NA*	RB *NA*	YCT *NA*	0
TEPEE CR							
8/01/1995	10S02W 13[1]	5	E	WCT 100	RB 0	YCT 0	5
7/28/ 1998	Km 1.6	13	P	WCT 98	RB 0	YCT 2	0
WALL CR							
7/13/1999	Km 5.6	7	P	WCT 97	RB 0	YCT 3	0
WIGWAM CR							
7/20/1999	08S02W 07[2]	7	P	WCT 82	RB 1	YCT 17	0

1/ Information from letter to Jim Brammer, Montana Fish Wildife and Parks (MFWP), from Robb Leary, University of Montana Wild Trout and Salmon Genetics Laboratory (WTSL) dated May 6, 1997.

2/ Information from letter from Naohisa Kanda, WTSL, to Brad Shepard, MFWP, dated March 27, 2000. Samples from locations in Buffalo Creek combined. An individual trout collected from Arasta Creek possessed PINE markers characteristic of both westslope and Yellowstone cutthroat trout.

3/ Information from letter to Brad Shepard, MFWP, from Naohisa Kanda, WTSL, dated August 21, 2000. A single allele characteristic of rainbow trout was present in one fish from Cabin Creek at T 11, R S04E, SEC 14, indicating either slight genetic introgression or a pure westslope cutthroat trout with a single deviant allele similar to rainbow trout.

4/ Information from letter to Brad Shepard, MFWP, from Naohisa Kanda, WTSL, dated November 8, 1999.

5/ In English George Creek a single allele characteristic of rainbow trout was present at low frequencies. This could indicate a small amount of hybridization or it could simply be a rare westslope cutthroat trout genetic variation. Information from letter from Naohisa Kanda, WTSL, to Brad Shepard, MFWP, dated March 27, 2000.

6/ Within the Horse Creek drainage (Horse and Tepee creeks) all fish were hybridized between westslope cutthroat, Yellowstone cutthroat, and rainbow trout, however, the population above a waterfall near stream mile 7.5 did not contain any rainbow trout alleles, had what may have been a few pure westslope cutthroat trout individuals, and had a higher proportion of westslope cutthroat trout alleles than the population below the falls. Information from letter to Brad Shepard, MFWP, from Naohisa Kanda, WTSL, dated November 8, 1999.

7/ Information from letter to Jim Brammer, MFWP, from Robb Leary, WTSL, dated May 23, 1995.

8/ Information from letter from Naohisa Kanda, WTSL, to Brad Shepard, MFWP, dated March 27, 2000. Proportions not available. Some fish that were pure WCT (6 of 9) were sampled at 4.0, 4.8, and 5.6 km. All fish below 4.0 km were either rainbow (3 of 15) or hybrids. The three hybrids above 4.0 km contained a single allele characteristic of RB.

9/ In the South Fork English George Creek, a single allele characteristic of Yellowstone cutthroat trout was present in one individual. It may be a pure westslope cutthroat trout population with a single deviant allele that is similar to Yellowstone cutthroat trout. Additional sampling is necessary. Information from letter from Naohisa Kanda, WTSL, to Brad Shepard, MFWP, dated March 27, 2000.

10/ Fish from the South Fork Indian Creek were all classified as hybrids between westslope cutthroat, rainbow, and Yellowstone cutthroat trout. However, fish from stream kilometer 4.0 to 5.6 contained over 90% westslope cutthroat trout alleles, while fish from lower in the drainage contained much lower westslope cutthroat trout allele frequencies. Information from letter to Brad Shepard, MFWP, from Naohisa Kanda, WTSL, dated November 8, 1999.

11/ Information from letter to Brad Shepard, MFWP, from Naohisa Kanda and Robb Leary, WTSL, dated November 2, 1998. A freezer malfunction made it impossible to detect differences between westslope cutthroat trout and rainbow trout alleles, so these populations should be sampled again. However, while westslope cutthroat trout genes were dominant, some Yellowstone cutthroat trout introgression was documented.

APPENDIX B

SALMONID RELATIVE ABUNDANCES IN MADISON RIVER TRIBUTARIES

Appendix B. Relative catches of westslope cutthroat (012), rainbow (001), brown (004), Yellowstone cutthroat (013), and hybrid(011) trout in the first electrofishing pass at all sampled sections in the Madison River drainage by stream, kilometer, date, species, and size class. The type of estimator, section length and width, and number of fish 75 mm and longer per 100 m are also presented.

DRAINAGE STREAM Kilometer	Date	Estimator	Section Length (m)	 Width (m)	Species	Number captured in first pass < 75 mm	 75-149 mm	 >150 mm	> 75 mm per 100 m in Pass 1
BEAR CREEK									
CAMERON CR									
0.1	6/29/98	1	43	0.3	No fish captured				0
M FK BEAR CR									
0.8	7/21/98	1	120	3.5	011	0	2	2	3
1.6	7/21/98	1	116	4.5	011	0	0	1	1
2.4	7/21/98	1	120	3	No fish captured				0
3.2	7/21/98	1	115	4.5	No fish captured				0
MILL CR									
5.6	7/20/98	1	75	3.5	No fish captured				0
6.4	7/20/98	1	110	3.2	No fish captured				0
N FK BEAR CR									
1.6	8/3/98	3	142	4	011	1	4	11	11
2.4	8/3/98	1	120	4	011	0	1	15	13
3.2	8/4/98	1	110	4	No fish captured				0
4.8	7/22/98	1	150	4	No fish captured				0
5.6	7/22/98	1	120	4	No fish captured				0
SHELL CR									
2.4	7/6/98	1	75	1.5	No fish captured				0
TOLMAN CR									
5.6	7/13/98	1	82	2.8	No fish captured				0

DRAINAGE STREAM Kilometer	Date	Estimator	Section		Species	Number captured in first pass			> 75 mm per 100 m in Pass 1
			Length (m)	Width (m)		< 75 mm	75-149 mm	>150 mm	
TRAIL FK BEAR CR									
0	7/7/98	1	80	1.9	No fish captured				0
0.8	7/14/98	1	258	2.8	012	0	0	4	2
1.6	7/7/98	1	110	2.6	No fish captured				0
2.4	7/7/98	1	50	2	No fish captured				0
UNNAMED TRIB TO BURGER CR									
1.6	6/30/98	1	76	1.2	No fish captured				0
CABIN CREEK									
CABIN CR									
1.8	7/26/99	1	944.88	9	012	0	2	5	1
2.6	7/26/99	1	457.2		012	0	0	11	2
3.1	7/26/99	1	255	9	012	0	0	1	1
3.5	7/26/99	1	140	9	012	0	2	3	4
4.3	7/26/99	1	120	7	012	0	5	7	10
6.6	7/28/99	1	100	7	012	0	0	2	2
9.3	7/27/99	1	160	6.6	012	0	0	3	2
9.4	7/27/99	1	31.09	6.6	012	0	1	3	13
9.5	7/27/99	1	87	1.5	012	0	0	1	1
9.7	7/27/99	1	420		012	0	2	6	2
GULLY CR									
0.1	7/27/99	1	150	2.5	012	0		2	1
M FK CABIN CR									
1.5	7/27/99	1	91	3.3	012	0	3	9	13
2.6	7/27/99	2	50	5.5	012	0	9	11	40
3.7	7/27/99	1	100	3.1	012	1	10	5	16

DRAINAGE STREAM Kilometer	Date	Estimator	Section Length (m)	 Width (m)	Species	Number captured in first pass < 75 mm	 75-149 mm	 >150 mm	> 75 mm per 100 m in Pass 1
4.4	7/27/99	1	50	3.8	012	0	6	5	22
5	7/28/99	2	131	2	012	0	3	12	12
6.9	7/28/99	2	146	1.5	012	0	10	28	26
CORRAL CREEK									
CORRAL CR									
9.7	7/8/98	3	93	2	011	2	3	4	10
10.5	7/8/98	1	75	2.5	No fish captured				0
ENGLISH GEORGE CREEK									
ENGLISH GEORGE CR									
0.8	6/23/97	2	107	0.6	004	0	4	1	5
0.8	6/23/97	2	107	0.6	001	0	1	0	1
1.1	6/23/97	1	100	0.6	012	0	1	1	2
1.6	7/1/97	1	50	0.4	No fish captured				0
2.4	7/1/97	1	100	0.6	No fish captured				0
3.2	6/23/97	1	50	0.8	012	0	1	0	2
4	6/23/97	1	50	0.7	012	3	6	0	18
4.3	6/26/97	3	103	0.8	012	5	21	2	27
4.8	6/23/97	1	50	0.9	012	0	2	1	6
S FK ENGLISH GEORGE CR									
0.3	7/1/97	1	100	0.7	No fish captured				0
0.8	7/1/97	1	100	0.7	No fish captured				0
1.6	6/25/97	1	50	0.5	012	0	2	1	6
1.8	7/1/97	1	50	0.7	012	0	2	3	10
2.4	6/26/97	3	100	1	012	0	3	8	11
3.2	6/25/97	1	50	1.4	012	1	5	3	18

DRAINAGE STREAM Kilometer	Date	Estimator	Section Length (m)	 Width (m)	Species	Number captured in first pass < 75 mm	 75-149 mm	 >150 mm	> 75 mm per 100 m in Pass 1
4	6/25/97	1	50	0.8	012	0	7	0	14
4.8	7/1/97	1	50	0.7	No fish captured				0
5	7/1/97	1	34	0.6	012	0	1	2	9
5.1	7/1/97	1	50	0.6	No fish captured				0
HORSE CREEK									
ALPINE CR									
0.2	7/16/97	1	50	0.6	No fish captured				0
0.6	7/16/97	1	50	1	No fish captured				0
HORSE CR									
4	8/13/97	1	50	5.5	011	0	1	17	36
4	8/13/97	1	50	5.5	004	0	0	1	2
4.8	8/13/97	1	100	5.1	011	0	5	16	21
7.2	7/30/98	1	100		011	0	0	5	5
8	7/30/98	1	100		011	0	0	7	7
8.8	7/30/98	3	104	5.2	011	0	1	15	15
9.3	7/30/98	1	104		011	1	2	8	11
10.5	7/30/98	1	125		011	0	3	5	6
11.3	7/30/98	2	112	5	011	0	2	12	12
12.1	7/27/98	2	120	5	012	0	0	6	5
12.9	7/28/98	2	112	4.3	012	0	1	2	3
13.7	7/28/98	3	105	4	012	4	6	10	19
14.5	7/29/98	1	102		012	5	0	6	11
15.3	7/28/98	2	115	2.5	012	0	0	11	10
16.1	7/16/97	3	102	2.3	012	0	4	24	28
16.9	7/17/97	1	50	0.7	012	0	0	4	8

DRAINAGE STREAM Kilometer	Date	Estimator	Section		Species	Number captured in first pass			> 75 mm per 100 m in Pass 1
			Length (m)	Width (m)		< 75 mm	75-149 mm	>150 mm	
17.7	7/17/97	1	50	0.8	No fish captured				0
TEPEE CR									
0.2	7/17/97	3	100	2	012	0	6	16	22
1.6	7/28/98	2	104	2.1	012	0	3	20	22
2.1	7/17/97	1	50	1.4	012	0	1	5	12
2.4	7/28/98	1	110		No fish captured				0
UNNAMED TRIB TO HORSE CR									
0.2	7/28/98	1	52		012	0	0	1	2
HYDE CREEK									
HYDE CR									
0.3	7/3/97	1	55	0.8	004	0	1	0	2
0.8	7/3/97	1	55	1	004	0	2	0	4
1.6	7/3/97	1	50	0.8	No fish captured				0
2.3	7/3/97	1	54	1.5	No fish captured				0
2.6	7/3/97	1	52	1	004	0	0	1	2
3.2	7/3/97	1	52	3.1	No fish captured				0
3.9	7/14/99	1	53		004	0	0	2	4
3.9	7/14/99	1	53	2.8	012	0	0	2	4
4.2	7/3/97	1	50	2.2	012	4	4	1	18
4.8	7/2/97	1	50	1.5	012	0	4	4	16
5.6	7/14/99	1	97	2	012	0	10	12	23
6.4	7/2/97	1	100	1.4	012	0	1	4	5
7.2	7/2/97	1	51	2	No fish captured				0
7.4	7/2/97	1	15	2.1	No fish captured				0
8	7/2/97	1	88	2.4	No fish captured				0

DRAINAGE STREAM Kilometer	Date	Estimator	Section		Species	Number captured in first pass			> 75 mm per 100 m in Pass 1
			Length (m)	Width (m)		< 75 mm	75-149 mm	>150 mm	
S FK HYDE CR									
0.4	7/2/97	1	50	1.6	No fish captured				0
INDIAN CREEK									
CIRCLE CR									
0.8	7/28/98	2	100	2.5	001	0	4	6	10
1.6	7/28/98	1	85	1.5	No fish captured				0
COUGAR CR									
0	7/27/98	1	100	1	No fish captured				0
0.8	7/27/98	1	105	3	No fish captured				0
GORGE CR									
0.8	7/29/98	2	120	2	001	0	0	4	3
1.6	7/29/98	1	95	1.5	No fish captured				0
INDIAN CR									
21.7	7/26/98	1	100	3	001	1	2	2	5
22.5	7/26/98	3	100	3	001	0	1	9	10
23.3	7/26/98	1	100	3	001	0	1	8	9
24.1	7/26/98	1	110	3.5	001	0	0	9	8
24.9	7/26/98	1	100	2.6	001	0	0	3	3
25.7	7/26/98	2	92	2.1	001	0	0	3	3
MANLEY CR									
0.8	7/28/98	1	75	1.5	No fish captured				0
1.6	7/28/98	1	85	1	No fish captured				0
MCATEE CR									
0.8	7/28/98	2	130	4.5	001	0	3	16	15
1.6	7/27/98	1	130	3.5	001	0	0	7	5
2.4	7/27/98	1	110	3.5	001	0	5	4	8

DRAINAGE STREAM Kilometer	Date	Estimator	Section Length (m)	 Width (m)	Species	Number captured in first pass < 75 mm	 75-149 mm	 >150 mm	> 75 mm per 100 m in Pass 1
3.2	7/27/98	2	100	3	001	0	0	12	12
4	7/27/98	1	98	3	No fish captured				0
NO MAN CR									
0.8	8/17/98	1	100	3	013	0	0	3	3
1.6	8/17/98	1	110	2.5	No fish captured				0
2.4	8/17/98	1	110	1.5	013	0	0	4	4
3.2	8/17/98	1	110	1.5	No fish captured				0
4	8/17/98	2	100	2	013	0	0	1	1
4.8	8/18/98	1	100	2.5	013	0	1	9	10
5.6	8/18/98	1	110	3	013	0	0	12	11
6.4	8/19/98	2	120	2.5	013	0	0	5	4
7.2	8/19/98	1	275	3	No fish captured				0
NO MAN LAKE OUTLET									
0.8	8/19/98	1		1	No fish captured				0
RAW LIVER CR									
0	7/26/98	1	60	0.7	No fish captured				0
S FK INDIAN CR									
1.6	8/4/98	1	120	3	011	1	1	5	6
2.4	8/6/98	3	120	4	011	0	6	10	13
3.2	8/4/98	1	105	3.1	011	0	1	4	5
4	8/5/98	1	100	2.5	012	0	0	3	3
4.8	8/5/98	1	110	2.5	012	0	3	7	9
5.6	8/5/98	1	100	2.5	012	0	0	4	4
7.2	8/5/98	1	100	3.5	No fish captured				0
8	8/5/98	1	75	2	No fish captured				0

DRAINAGE STREAM Kilometer	Date	Estimator	Section Length (m)	Section Width (m)	Species	Number captured in first pass < 75 mm	Number captured in first pass 75-149 mm	Number captured in first pass >150 mm	> 75 mm per 100 m in Pass 1
SHEDHORN CR									
0.8	7/14/98	1	120	3.8	No fish captured				0
1.6	7/14/98	1	100	2.5	No fish captured				0
UNNAMED TRIBUTARY									
0.4	7/29/98	1	80	1	001	0	0	1	1
0.8	8/19/98	1	75	1	No fish captured				0
1.6	7/29/98	1	75	0.5	No fish captured				0
LAKE CREEK									
LAKE CR									
1.8	6/25/98	3	102		004	0	54	7	60
2.6	6/25/98	3	100		004	0	66	18	84
3.4	6/24/98	3	120	6	004	22	7	8	31
MOOSE CREEK									
BAD LUCK CR									
4	7/8/98	1	70	1	No fish captured				0
MOOSE CR									
0.4	7/19/99	4	120	4.2	004	5	20	8	28
1.2	7/19/99	1	100	2.7	004	3	18	10	31
2	7/19/99	1	95	3.3	004	0	21	13	36
2.8	7/20/99	1	100	2.8	004	17	30	14	61
3.6	7/20/99	3	94	2.1	004	1	1	7	10
4.4	7/20/99	1	90	2	004	0	0	1	1
5.2	7/20/99	1	93	2	004	0	5	4	10
6	7/20/99	1	75	2.4	No fish captured				0
6.8	7/20/99	1	80	2	No fish captured				0

DRAINAGE STREAM Kilometer	Date	Estimator	Section Length (m)	 Width (m)	Species	Number captured in first pass < 75 mm	 75-149 mm	 >150 mm	> 75 mm per 100 m in Pass 1
7.6	7/21/99	1	85	2.6	No fish captured				0
8.8	7/21/99	1	90	2	No fish captured				0
10.1	7/21/99	1	75	2.3	No fish captured				0
10.9	7/21/99	1	90	2.3	013	0		2	2
11.7	7/21/99	2	140	3	013	0	8	42	36
UNNAMED TRIB TO MOOSE #1									
0.4	8/17/99	1	75	0.5	004	2	2	3	9
UNNAMED TRIB TO MOOSE #2									
0.8	8/17/99	1	85	1	004	2	9	8	22
NICKERSON CREEK									
NICKERSON CR									
3.1	7/9/97	1	110	0.7	No fish captured				0
3.4	7/9/97	1			No fish captured				0
S FK NICKERSON CR									
0.2	7/9/97	1			No fish captured				0
PAPOOSE CREEK									
PAPOOSE CR									
0.8	7/29/99	2	124	4.2	004	2	6	3	9
0.8	7/29/99	2	124	4.2	011	0	3	1	4
0.8	7/29/99	2	124	4.2	001	0	0	1	1
1.6	7/26/99	1	100	3.2	004	1	0	0	1
1.6	7/26/99	1	100	3.2	011	1	5	2	8
2.4	7/26/99	1	95	3.5	011	0	3	1	4
2.4	7/26/99	1	95	3.5	004	0	0	6	6
3.2	7/26/99	1	120	4	004	0	0	2	2

DRAINAGE STREAM Kilometer	Date	Estimator	Section		Species	Number captured in first pass			> 75 mm per 100 m in Pass 1
			Length (m)	Width (m)		< 75 mm	75-149 mm	>150 mm	
3.2	7/26/99	1	120	4	011	0	1	1	2
4	7/26/99	1	110	3	012	0	1	0	1
4.8	7/26/99	4	125	3.2	012	0	2	2	3
5.6	7/27/99	1	100	3.5	012	0	0	3	3
6.4	7/27/99	1	115	4	No fish captured				0
7.2	7/27/99	0		4	No fish captured				0
8	7/27/99	1	117	4	No fish captured				0
9.3	7/28/99	1	95	4	No fish captured				0
10.1	7/28/99	1	110	3	No fish captured				0
UNNAMED TRIBUTARY									
0.4	7/28/99	1	90	2.8	No fish captured				0
QUAKING ASPEN CREEK									
QUAKING ASPEN CR									
0.8	6/30/98	1	62	1.9	001	0	5	0	8
0.8	6/30/98	1	62	1.9	011	0	5	0	8
2.6	6/30/98	2	85	2	001	0	1	0	1
2.6	6/30/98	2	85	2	011	0	10	2	14
3.4	6/30/98	1	85	2	011	0	4	0	5
3.9	6/30/98	1	75	1.5	No fish captured				0
4.2	6/30/98	1	100	1.5	No fish captured				0
RUBY CREEK									
RUBY CR									
0.8	7/10/97	1	102	2.1	001	0	0	1	1
0.8	7/10/97	1	102	2.1	004	3	14	5	22
1.6	7/10/97	1	100	2.8	001	0	1	8	9

DRAINAGE STREAM Kilometer	Date	Estimator	Section		Species	Number captured in first pass			> 75 mm per 100 m in Pass 1
			Length (m)	Width (m)		< 75 mm	75-149 mm	>150 mm	
4	7/10/97	1	100	3.5	001	0	5	30	35
5	6/29/98	1	75	6.5	001	0	3	1	5
10.5	7/13/98	1	10	5	001	0	7	10	170
12.9	7/13/98	1	75	3.5	No fish captured				0
16.1	7/13/98	1	75	3.5	No fish captured				0
17.7	7/13/98	1	75	3	No fish captured				0
20.1	7/13/98	1	75	3	No fish captured				0
21.2	7/13/98	1	100	2.3	No fish captured				0
SOAP CREEK									
SOAP CR									
2	7/20/95	2	70	3.5	012	1	18	3	31
2.5	7/20/95	2	83	3	012	6	17	0	28
3.9	7/20/95	2	135	2.5	012	9	11	3	17
5.4	7/20/95	3	127	2.5	012	0	4	4	6
SQUAW CREEK									
MIDDLE FORK SQUAW CR									
0.8	8/5/99	1	115	3.6	No fish captured				0
1.6	8/5/99	1	98	3	013	0	0	1	1
2.4	8/5/99	1	163	3.8	No fish captured				0
3.2	8/5/99	1	105	3.5	013	0	0	2	2
4	8/11/99	1	105	2.8	No fish captured				0
4.8	8/11/99	1	103	3	013	0	0	1	1
5.6	8/11/99	1	90	3	No fish captured				0
6.4	8/11/99	1	93	3.1	No fish captured				0

DRAINAGE STREAM Kilometer	Date	Estimator	Section Length (m)	Section Width (m)	Species	Number captured in first pass < 75 mm	Number captured in first pass 75-149 mm	Number captured in first pass >150 mm	> 75 mm per 100 m in Pass 1
NORTH FORK SQUAW CR									
0.3	8/12/99	1	83.	2.5	011	0	0	2	2
0.8	8/10/99	1	75	2	No fish captured				0
SOUTH FORK SQUAW CR									
1.2	8/4/99	1	75	1.5	004	1	9	0	13
SQUAW CR									
0.8	8/9/99	1	131	4.5	011	0	0	1	1
0.8	8/9/99	1	131	4.5	004	0	13	9	17
0.8	8/9/99	1	131	4.5	001	0	0	2	2
1.6	8/9/99	1	128	4.5	001	0	1	2	2
1.6	8/9/99	1	128	4.5	011	0	0	1	1
1.6	8/9/99	1	128	4.5	004	2	9	13	19
3.2	8/9/99	2	125	4	011	0	0	2	2
3.2	8/9/99	2	125	4	004	0	5	25	24
4.8	8/9/99	1	118	4	011	0	6	16	19
4.8	8/9/99	1	118	4	004	0	0	1	1
5.6	8/11/99	1	123	4	011	0	1	3	3
6.4	8/12/99	2	121	4.5	011	0	0	2	2
8	8/12/99	1	115	4	011	1	0	0	1
9.7	8/10/99	1	98	2	No fish captured				0
11.3	8/10/99	1	83	2	011	0	1	2	4
12.9	8/10/99	2	85	2	011	3	6	6	18
13.5	8/10/99	1	70	1	013	0	1	1	3

DRAINAGE STREAM Kilometer	Date	Estimator	Section Length (m)	Section Width (m)	Species	Number captured in first pass < 75 mm	Number captured in first pass 75-149 mm	Number captured in first pass >150 mm	> 75 mm per 100 m in Pass 1
STANDARD CREEK									
STANDARD CR									
0.8	8/11/97	1	50	12.5	004	0	1	0	2
0.8	8/11/97	1	50	12.5	001	1	7	7	30
1.6	8/11/97	1	50	12	001	1	10	8	38
1.6	8/11/97	1	50	12	012	2	1	2	10
2.4	7/30/97	1	50	4.3	012	1	3	3	14
3.2	7/30/97	1	50	5.8	012	1	1	4	12
4	7/30/97	1	50	5.6	012	1	4	9	28
4.8	7/30/97	1	50	9.4	012	1	0	9	20
5.6	7/29/97	1	50	9	012	0	2	2	8
6.4	7/30/97	1	50	7.5	012	0	1	5	12
7.2	7/30/97	1	50	6.8	012	0	2	2	8
8	7/29/97	1	55	4.8	012	0	0	3	6
8.8	7/29/97	1	51	6	No fish captured				0
9.7	7/29/97	2	50	5.5	012	0	0	14	28
10.5	7/29/97	1	52	4.6	012	0	1	6	14
11.3	7/29/97	1	57	4.4	No fish captured				0
12.1	7/29/97	1	53	4.2	012	0	0	1	2
13.7	7/29/97	1	56	3.5	012	0	0	2	4
14.5	7/29/97	1	60	3.6	No fish captured				0
16.1	7/28/97	1	52	4	No fish captured				0
16.9	7/28/97	1	61	3.1	No fish captured				0
17.7	7/28/97	1	53	1.4	No fish captured				0
18.5	7/28/97	1	53	1.6	No fish captured				0

DRAINAGE STREAM Kilometer	Date	Estimator	Section Length (m)	Section Width (m)	Species	Number captured in first pass < 75 mm	Number captured in first pass 75-149 mm	Number captured in first pass >150 mm	> 75 mm per 100 m in Pass 1
19.3	7/28/97	1	53	1.7.	No fish captured				0
WOLVERINE CR									
0.6	7/30/97	1	52	4.2	No fish captured				0
1.6	7/30/97	1	300	2.6	011	0	1	6	2
2.4	7/30/97	1	83	2.4	013	0	1	1	2
3.2	7/30/97	1	108	1	No fish captured				0
THREE DOLLAR SPRING									
$3 BRIDGE SPRING CR									
0.2	6/18/98	1	157	2.2	004	0	10	1	7
0.3	6/18/98	1	152	6.2	004	63	39	11	74
WALL CREEK									
N FK WALL CR									
0.2	7/9/97	1	50	1.7	012	2	3	1	12
0.5	7/9/97	1	50	0.8	012	8	1	0	18
0.8	7/9/97	1	50	1	No fish captured				0
WALL CR									
0.8	7/10/97	1	50	1.8	004	0	3		6
0.8	7/10/97	1	50	1.8	001	0	8	1	18
1.6	7/10/97	1	50	2.3	001	0	5	1	12
2.4	7/10/97	1	50	2.2	001	0	0	2	4
3.2	7/9/97	1	50	2.1	No fish captured				0
4	7/9/97	1	51	2.6	012	5	2	5	24
4.8	7/8/97	3	100	1.9	012	1	6	8	15
5.6	7/9/97	1	52	2.8	012	0	7	8	29
6.4	7/9/97	1	50	2.6	012	0	7	4	22

DRAINAGE STREAM Kilometer	Date	Estimator	Section Length (m)	Section Width (m)	Species	Number captured in first pass < 75 mm	Number captured in first pass 75-149 mm	Number captured in first pass >150 mm	> 75 mm per 100 m in Pass 1
7.2	7/9/97	1	50	2.4	012	0	3	5	16
8	7/9/97	1	50	1	No fish captured				0
8.8	7/9/97	1	50	1.2	No fish captured				0
9.7	7/9/97	1	50	2	No fish captured				0
WIGWAM CREEK									
ARASTA CR									
0.8	7/22/97	1	50	2	012	0	0	1	2
1.6	7/23/97	1	57	2.6	012	2	1	1	7
2.4	7/23/97	1	49	1.4	012	0	2	0	4
3.2	7/23/97	1	58	1.5	012	1	0	0	2
4	7/23/97	1	50	1.4	No fish captured				0
BUFFALO CR									
0.8	7/22/97	1	53	2	011	2	14	1	32
1.6	7/22/97	1	49	2.8	No fish captured				0
2.4	7/22/97	2	100	1.9	011	0	5	3	8
3.2	7/22/97	1	53	1.5	011	1	1	0	4
4	7/23/97	1	55	1.3	011	0	1	0	2
4.8	7/23/97	1	50	0.4	No fish captured				0
UNNAMED TRIB									
0.3	9/21/97	1	57	0.7	No fish captured				0
WIGWAM CR									
6.9	8/19/98	1	130		No fish captured				0
7.9	8/19/98	1	105	2.2	011	0	0	3	3
9.7	8/12/97	1	60	3.4	011	0	0	3	5
10.5	8/12/97	1	55	3	011	0	0	2	4

DRAINAGE									
STREAM			Section			Number captured in first pass			> 75 mm per 100 m in Pass 1
Kilometer	Date	Estimator	Length (m)	Width (m)	Species	< 75 mm	75-149 mm	>150 mm	
11.3	8/12/97	1	50	3.7	011	0	1	2	6
12.1	8/12/97	1	100	3.3	011	0	3	2	5
12.9	9/21/97	1	50	2.8	011	0	6	3	18
13.7	7/22/97	1	53	2.5	No fish captured				0
14.5	7/21/97	1	56	10.2	011	0	0	1	2
15.3	9/21/97	1	50	1.2	No fish captured				0
16.1	9/21/97	1	50	1.8	No fish captured				0
16.9	9/21/97	1	56	3.3	No fish captured				0
WOLF CREEK									
UNNAMED TRIBUTARY									
0.4	8/4/99	1	80	2	No fish captured				0
WOLF CR									
0.8	8/2/99	1	125	4	004	0	3	12	12
0.8	8/2/99	1	125	4	011	0	1	0	1
1.6	8/2/99	1	115	3.5	004	0	3	9	10
1.6	8/2/99	1	115	3.5	001	0	1	1	2
2.4	8/2/99	1	120	3.5	001	0	0	2	2
2.4	8/2/99	1	120	3.5	004	1	29	9	32
3.2	8/2/99	2	120	3	004	0	1	5	5
3.2	8/2/99	2	120	3	001	0	1	2	2
4	8/2/99	1	120	3.5	004	0	0	1	1
4.8	8/3/99	1	124	4	001	0	0	4	3
5.6	8/3/99	1	119	4	001	0	0	3	2
5.6	8/3/99	1	119	4	011	0	0	1	1
6.4	8/3/99	2	123	4.5	001	0	0	3	2

DRAINAGE STREAM Kilometer	Date	Estimator	Section Length (m)	 Width (m)	Species	Number captured in first pass < 75 mm	 75-149 mm	 >150 mm	> 75 mm per 100 m in Pass 1
7.2	8/4/99	1	125	5	011	0	0	2	2
8.8	8/4/99	1	123	5	No fish captured				0
10.5	8/4/99	1	150	4	No fish captured				0

APPENDIX C

ESTIMATED NUMBERS OF SALMONID SPECIES IN MADISON RIVER TRIBUTARIES

Appendix C. Estimated numbers and standard errors (SE) of westslope cutthroat (012), rainbow (001), brown (004), Yellowstone cutthroat (013), and hybrid (011) trout in sampled sections in the Madison River drainage by stream, kilometer, date, species, and size class. The type of estimator, section length and width, and estimated number of fish 75 mm and longer per 100 m are also presented.

DRAINAGE STREAM Kilometer	Date	Section length (m)	Species	Estimate (SE) by length group <75 mm	 75-149 mm	 >150 mm	Length range (mm) Min	 Max	Total estimate (SE)	Estimated number per 100 m (SE)
BEAR CREEK										
N FK BEAR CR										
1.6	8/3/98	142	011	1 (0)	9 (2.6)	17 (0.7)	69	351	27 (1.7)	19 (1.2)
CABIN CREEK										
M FK CABIN CR										
2.6	7/27/99	50	012	0 (0)	9 (0)	0 (0)	97	243	22 (0.5)	44 (0.9)
5	7/28/99	131	012	0 (0)	4 (0.6)	0 (0)	100	211	18 (0.8)	13.7 (0.6)
6.9	7/28/99	146	012	0 (0)	12 (0.7)	0 (0)	92	258	44 (1)	30.1 (0.7)
ENGLISH GEORGE CREEK										
ENGLISH GEORGE CR										
0.8	6/23/97	107	004	0 (0)	6 (0.5)	0 (0)	107	152	6 (0.5)	5.6 (0.4)
4.3	6/26/97	103	012	0 (0)	34 (1.9)	2 (0)	43	165	36 (1.8)	35 (1.8)

DRAINAGE STREAM Kilometer	Date	Section length (m)	Species	Estimate (SE) by length group <75 mm	 75-149 mm	 >150 mm	Length range (mm) Min	 Max	Total estimate (SE)	Estimated number per 100 m (SE)
S FK ENGLISH GEORGE CR										
2.4	6/26/97	100	012	0 (0)	6 (0.7)	10 (0.2)	109	226	16 (0.6)	16 (0.6)
HORSE CREEK										
HORSE CR										
8.8	7/30/98	104	011	0 (0)	2 (0.4)	22 (1)	97	300	24 (1)	23.1 (1)
11.3	7/30/98	112	011	0 (0)	2 (0)	14 (0.6)	124	272	16 (0.6)	14.3 (0.5)
12.1	7/27/98	120	012	0 (0)	0 (0)	7 (0.5)	51	320	7 (0.4)	5.8 (0.4)
12.9	7/28/98	112	012	0 (0)	1 (0)	2 (0)	109	239	3 (0)	2.7 (0)
13.7	7/28/98	105	012	7 (1)	10 (0.9)	13 (0.5)	76	267	31 (1.8)	29.5 (1.7)
15.3	7/28/98	115	012	0 (0)	0 (0)	12 (0.3)	155	297	12 (0.3)	10.4 (0.3)
16.1	7/16/97	102	012	0 (0)	8 (0.8)	29 (0.4)	104	272	37 (0.7)	36.3 (0.7)
TEPEE CR										
0.2	7/17/97	100	012	0 (0)	11 (1)	18 (0.1)	94	241	29 (0.6)	29 (0.6)

DRAINAGE STREAM Kilometer	Date	Section length (m)	Species	Estimate (SE) by length group <75 mm	 75-149 mm	 >150 mm	Length range (mm) Min	 Max	Total estimate (SE)	Estimated number per 100 m (SE)
1.6	7/28/98	104	012	0 (0)	4 (0.6)	23 (0.7)	76	257	27 (0.9)	26 (0.8)
HYDE CREEK										
HYDE CR										
5.6	7/2/97	101	012	0 (0)	18 (0.5)	19 (0.8)	104	216	37 (0.9)	36.6 (0.9)
INDIAN CREEK										
CIRCLE CR										
0.8	7/28/98	100	001	0 (0)	6 (1.1)	6 (0)	109	211	12 (0.7)	12 (0.7)
CORRAL CR										
9.7	7/8/98	93	011	9 (1)	8 (5.7)	6 (0.4)	53	201	24 (4.1)	25.8 (4.4)
GORGE CR										
0.8	7/29/98	120	001	0 (0)	0 (0)	4 (0)	165	206	4 (0)	3.3 (0)
INDIAN CR										
22.5	7/26/98	100	001	1 (0)	3 (1.8)	10 (0.1)	56	239	14 (0.5)	14 (0.5)
25.7	7/26/98	92	001	0 (0)	0 (0)	3 (0)	178	211	3 (0)	3.3 (0)

DRAINAGE STREAM Kilometer	Date	Section length (m)	Species	Estimate (SE) by length group <75 mm	 75-149 mm	 >150 mm	Length range (mm) Min	 Max	Total estimate (SE)	Estimated number per 100 m (SE)
MCATEE CR										
0.8	7/28/98	130	001	0 (0)	3 (0)	18 (0.5)	109	264	21 (0.5)	16.2 (0.4)
3.2	7/27/98	100	001	0 (0)	0 (0)	12 (0)	180	279	12 (0)	12 (0)
NO MAN CR										
4	8/17/98	100	013	0 (0)	0 (0)	1 (0)	196	196	1 (0)	1 (0)
6.4	8/19/98	120	013	0 (0)	0 (0)	6 (0.5)	262	305	6 (0.5)	5 (0.4)
S FK INDIAN CR										
2.4	8/6/98	120	011	0 (0)	9 (0.5)	13 (0.5)	76	249	22 (0.7)	18.3 (0.6)
LAKE CREEK										
LAKE CR										
1.8	6/25/98	102	004	0 (0)	176 (37.6)	16 (2)	91	277	189 (32.9)	185.3 (32)
2.6	6/25/98	100	004	0 (0)	128 (7.5)	35 (5)	89	269	164 (9.4)	164 (9.4)
3.4	6/24/98	120	004	0 (0)	18 (4.7)	14 (2)	76	353	35 (5.4)	29.2 (4.5)

DRAINAGE STREAM Kilometer	Date	Section length (m)	Species	Estimate (SE) by length group			Length range (mm)		Total estimate (SE)	Estimated number per 100 m (SE)
				<75 mm	75-149 mm	>150 mm	Min	Max		
MOOSE CREEK										
MOOSE CR										
0.4	7/19/99	120	004	20 (6)	47 (3.1)	13 (0.5)	28	251	60 (2.9)	50 (2.4)
3.6	7/20/99	94	004	2 (0)	6 (4.7)	10 (0.6)	71	284	16 (2.1)	17 (2.3)
11.7	7/21/99	140	013	0 (0)	11 (1.1)	52 (2)	94	239	65 (2.9)	46.4 (2.1)
PAPOOSE CREEK										
PAPOOSE CR										
0.8	7/29/99	124	011	0 (0)	3 (0)	1 (0)	109	193	4 (0)	3.2 (0)
0.8	7/29/99	124	004	0 (0)	0 (0)	0 (0)	64	196	9 (0.4)	7.3 (0.3)
0.8	7/29/99	124	001	0 (0)	0 (0)	2 (1)	338	396	2 (1)	1.6 (0.8)
6.4	7/27/99	115	012	0 (0)	6 (1)	2 (0)	89	253	8 (0.7)	7 (0.6)
QUAKING ASPEN CREEK										
QUAKING ASPEN CR										
2.6	6/30/98	85	011	0 (0)	14 (1)	2 (0)	84	165	16 (0.9)	18.8 (1.0)

DRAINAGE STREAM Kilometer	Date	Section length (m)	Species	Estimate (SE) by length group <75 mm	 75-149 mm	 >150 mm	Length range (mm) Min	 Max	Total estimate (SE)	Estimated number per 100 m (SE)
SOAP CREEK										
SOAP CR										
2	7/20/95	70	012	1 (0)	21 (0.7)	0 (0)	56	235	28 (2)	40 (2.9)
2.5	7/20/95	83	012	7 (0)	20 (0.8)	0 (0)	55	146	20 (0.8)	24.1 (1)
3.9	7/20/95	135	012	12 (0)	11 (0.3)	0 (0)	51	169	14 (0.6)	10.4 (0.4)
5.4	7/20/95	127	012	0 (0)	5 (0.5)	0 (0)	91	196	11 (1.1)	8.7 (0.9)
SQUAW CREEK										
MIDDLE FORK SQUAW CR										
4.8	8/11/99	103	013	0 (0)	0 (0)	1 (0)	330	330	1 (0)	1 (0)
SQUAW CR										
3.2	8/9/99	125	011	0 (0)	0 (0)	3 (0.7)	193	249	3 (0.8)	2.4 (0.6)
3.2	8/9/99	125	004	2 (0)	3 (0)	29 (0.6)	76	295	32 (0.6)	25.6 (0.5)
6.4	8/12/99	121	011	0 (0)	0 (0)	3 (0.7)	180	234	3 (0.8)	2.5 (0.6)
12.9	8/10/99	85	011	3 (1)	7 (0.4)	8 (0.9)	51	221	15 (0.9)	17.6 (1.1)

DRAINAGE STREAM Kilometer	Date	Section length (m)	Species	Estimate (SE) by length group			Length range (mm)		Total estimate (SE)	Estimated number per 100 m (SE)
				<75 mm	75-149 mm	>150 mm	Min	Max		
STANDARD CREEK										
STANDARD CR										
9.7	7/29/97	50	012	0 (0)	0 (0)	14 (0)	160	254	14 (0)	28 (0)
THREE DOLLAR SPRING - WEST										
$3 BRIDGE SPRING CR										
0.2	7/21/97	150	004	91 (55)	93 (40.9)	3 (0)	43	201	87 (28)	58 (19)
WALL CREEK										
WALL CR										
4.8	7/8/97	100	012	3 (1)	9 (0.5)	11 (0.6)	48	221	20 (0.7)	20 (0.7)
WIGWAM CREEK										
BUFFALO CR										
2.4	7/22/97	100	011	0 (0)	5 (0)	3 (0)	104	188	8 (0)	8 (0)
WOLF CREEK										
WOLF CR										
3.2	8/2/99	120	001	0 (0)	1 (0)	3 (0.5)	145	295	4 (0.6)	3.3 (0.5)
3.2	8/2/99	120	004	0 (0)	1 (0)	5 (0)	104	292	6 (0)	5 (0)

DRAINAGE

STREAM				Estimate (SE) by length group			Length range (mm)		Total	
Kilometer	Date	Section length (m)	Species	<75 mm	75-149 mm	>150 mm	Min	Max	estimate (SE)	Estimated number per 100 m (SE)
6.4	8/3/99	123	001	0 (0)	0 (0)	4 (0.6)	203	305	4 (0.6)	3.3 (0.5)