MONTANA STATE UNIVERSITY

Survey of habitat and fish communities in segments of Cherry Creek, a proposed site for the re-introduction of westslope cutthroat trout
by Sean Spence-Patrick Moran

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in Biological Sciences
Montana State University
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Abstract:
Cherry Creek has been identified as a site for restoration of westslope cutthroat trout. The purpose of this study was to provide pre-restoration baseline data to compare to the future restored westslope cutthroat trout population. These comparisons enabled by the fish community and habitat data will provide a means to judge the relative success of the proposed restoration effort. Three principal reaches were surveyed for physical habitat following R1/R4 stream habitat inventory procedures. These reaches encompassed the area of Cherry Creek between a lower barrier falls and the higher gradient headwaters. Fish population and demographic parameters were estimated from fish gathered during electrofishing depletion estimates. These estimates were performed on three monitoring sections (one in each principal reach) of 402 to 512 m in length during the late summer of 1998 and 1999. Fish population and demographic parameters estimated include: population, density, biomass, length frequency, percent-at-age, length-at-age, annual growth, condition factor, and annual mortality. Macrohabitat results indicated an abundance of high quality habitat, with low gradients of 1.0 to 1.5 %, slow water habitat comprising 20 to 32 % of stream length, mean pool maximum depth of about 1 meter, low sediment, an abundance of woody cover, and favorable water temperatures. Fish population estimates of 835 to 1285 fish/ km, combined biomass estimates of 6.99 to 15.24 g/m2, and combined density estimates of 0.105 to 0.170 fish/m2, for the three sections over the two years indicate a productive system. Length-frequency, and scale and otolith readings indicate a young population, with very few (8 of 247) fish 4 years of age or older. Brook trout comprised a greater percent of the population as the reaches progressed upstream. Mean length-at-age, and back-calculations indicated fast annual growth for the first 2 age classes and much slower growth for ages 2 and above for both species in all sections. Condition factors of about 1.1 for both species in all sections indicated better than average condition. Annual mortality rates were consistently high for both species in all sections over the two years, ranging from 46.0 % (95% CL +/- 39.4 %) to 91.4 %. Habitat and fish correlations indicated that gradient is negatively correlated with rainbow trout distribution and that pool volume is positively correlated with trout growth. Results of this study indicate that lower Cherry Creek has the ability to support a large salmonid population and should the restored westslope cutthroat trout population resemble the present trout community in terms of demographic parameters, the restoration should prove successful.

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Sean Spence-Patrick Moran

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

Cherry Creek has been identified as a site for restoration of westslope cutthroat trout. The purpose of this study was to provide pre-restoration baseline data to compare to the future restored westslope cutthroat trout population. These comparisons enabled by the fish community and habitat data will provide a means to judge the relative success of the proposed restoration effort. Three principal reaches were surveyed for physical habitat following R1/R4 stream habitat inventory procedures. These reaches encompassed the area of Cherry Creek between a lower barrier falls and the higher gradient headwaters. Fish population and demographic parameters were estimated from fish gathered during electrofishing depletion estimates. These estimates were performed on three monitoring sections (one in each principal reach) of 402 to 512 m in length during the late summer of 1998 and 1999. Fish population and demographic parameters estimated include: population, density, biomass, length frequency, percent-at-age, length-at-age, annual growth, condition factor, and annual mortality. Macrohabitat results indicated an abundance of high quality habitat, with low gradients of 1.0 to 1.5 %, slow water habitat comprising 20 to 32 % of stream length, mean pool maximum depth of about 1 meter, low sediment, an abundance of woody cover, and favorable water temperatures. Fish population estimates of 835 to 1285 fish/ km, combined biomass estimates of 6.99 to 15.24 g/m^2, and combined density estimates of 0.105 to 0.170 fish/m^2, for the three sections over the two years indicate a productive system. Length-frequency, and scale and otolith readings indicate a young population, with very few (8 of 247) fish 4 years of age or older. Brook trout comprised a greater percent of the population as the reaches progressed upstream. Mean length-at-age, and back-calculations indicated fast annual growth for the first 2 age classes and much slower growth for ages 2 and above for both species in all sections. Condition factors of about 1.1 for both species in all sections indicated better than average condition. Annual mortality rates were consistently high for both species in all sections over the two years, ranging from 46.0 % (95% CL +/- 39.4 %) to 91.4 %. Habitat and fish correlations indicated that gradient is negatively correlated with rainbow trout distribution and that pool volume is positively correlated with trout growth. Results of this study indicate that lower Cherry Creek has the ability to support a large salmonid population and should the restored westslope cutthroat trout population resemble the present trout community in terms of demographic parameters, the restoration should prove successful.

INTRODUCTION

Westslope cutthroat trout *(Oncorhynchus clarki lewisi)* have been petitioned for listing as a threatened species under the Federal Endangered Species Act (USFWS 2000). Furthermore, westslope cutthroat trout are classified as a State of Montana Class A species, where "limited numbers or limited habitat both in Montana and elsewhere in North America; elimination from Montana would be a significant loss to the gene pool of the species or subspecies" (Hunter 1994).

Habitat degradation, competition, and hybridization with non-native salmonids are the principle reasons why westslope cutthroat trout are now limited to approximately 2.5 percent of their historic range in the upper Missouri River (Shepard et al. 1997). As a result of this decline, the interagency Westslope Cutthroat Trout Steering Committee has set a goal of establishing five areas with at least fifty miles of interconnected habitat in the upper Missouri River basin (MFWP 1999). Cherry Creek has been identified as a project site as part of the Montana Fish, Wildlife and Parks (MFWP) Madison River Drainage Westslope Cutthroat Trout Conservation and Restoration Program (Bramblett 1998).

Unfortunately, this decline, its causes, and the subsequent management strategies employed are not limited to westslope cutthroat trout. The majority of inland salmonid species and subspecies of North America are facing similar circumstances (Behnke 1992). Restoration efforts have been instituted for many threatened species and subspecies (Gresswell 1988, Rinne and Turner 1991, Young 1995), notably: Yellowstone cutthroat trout *O. clarki bouvieri* (Thurow et al. 1988), Colorado River cutthroat trout *O. clarki pleuriticus* (U.S. Department of the Interior 1995), greenback cutthroat trout *O. clarki stomias* (Stuber et al. 1988, Harig et al. 2000), Lahontan cutthroat trout *O. clarki henshawi* (U.S. Fish and Wildlife Service 1995), Gila trout *O. gilae* (U.S. Fish and Wildlife Service 1993), Apache trout *O. apache* (Hanson and David 1989), golden trout *O. mykiss aguabonita* (Pister 1998), and Arctic grayling *Thymallus arcticus* (Kaya 1992, Magee 1998).

The overall management objective of recovery plans, whether they involve trout or other threatened species, is typically delisting or prevention of listing of the species or subspecies in peril (U.S. Fish and Wildlife Service 1995). A threatened or endangered trout species may be considered for delisting, or a candidate species left unlisted, when the restoration plan achieves the goal of increased distribution and abundance. This is in addition to the primary objective of protecting the remaining populations from further decline. Common management strategies employed to meet these goals and objectives include: preservation and rehabilitation of necessary habitat, changes in fishing regulations, establishment of genetically pure and diverse broodstocks, reintroduction of the species or subspecies to enhance its overall numbers and range, and monitoring of the restoration to evaluate success or need for further management (Stuber et al. 1988, Rinne and Turner 1991, U.S. Fish and Wildlife Service 1993, U.S. Fish and Wildlife Service 1995, Young 1995, Pister 1998, Harig et al. 2000).

In the majority of instances, successful restoration of cutthroat trout requires that established, non-native salmonid populations are completely removed and prevented from recolonizing the reintroduction area (Horan et al. 2000). The need for removal and separation from existing non-native trout is indicated by the extensive history of native cutthroat trout being competitively displaced by or becoming hybridized with, non-native trout (Allendorf and Leary 1988, Griffith 1988, Rinne and Turner 1991, Shepard et al. 1997). For cutthroat trout, hybridization with rainbow trout or other subspecies of cutthroat trout has been implicated as the major cause in elimination of numerous populations (Allendorf and Leary 1988, Gresswell 1988, Liknes and Graham 1988, Shepard et al. 1988). The few exceptions to this occur where there is spawning isolation (Liknes and Graham 1988, Thurow et al. 1988), or where suitable habitat and negligible or more regulated fishing pressure, have resulted in cutthroat trout populations that are able to persist with non-natives (Griffith 1988, Thurow et al. 1988, Young 1995). Also, areas of harsh conditions, such as cold, high gradient headwaters where a locally adapted subspecies may have a competitive advantage over an invading non-native, have allowed some cutthroat populations to persist (Gerstung 1988, Griffith 1988, Young 1995, U.S. Fish and Wildlife Service 2000).

It is hardly surprising then, that headwaters constitute the majority of the present distribution of endangered inland trout (Behnke 1992, U.S. Fish and Wildlife Service 2000). These smaller headwater streams also represent the best opportunity for increasing distributions of these species because they are more amenable to the intensive management needed for successful restoration. Paradoxically, the need to institute restoration projects on larger, lower elevation and more productive stream systems is widely recognized (Horan et al. 2000, Hilderbrand and Kershner 2000, Shepard and

Spoon 2001). These larger, more productive, and more physically diverse drainages provide the population numbers needed to ensure survival as well as insuring life history and genetic diversity (Allendorf and Leary 1988, Horan et al. 2000, Hilderbrand and Kershner 2000). Of additional importance is that when entire interconnected sub-basins such as Cherry Creek are used, the restored population is more resilient to local, stochastic events that could cause the extinction of a restored population from a more typical isolated headwater area (Probst et al. 1992, Hilderbrand and Kershner 2000, Horan et al. 2000). The selection of Cherry Creek as a restoration site addresses the desirability for restoration projects to incorporate systems larger and more productive than the headwater streams typically used in restoration projects.

Successful restoration requires the use of piscicides, such as rotenone and antimycin, to ensure the eradication of non-native species because of the unreliability of other means such as electrofishing (Moore et al.1983, U.S. Department of the Interior 1995, Thompson and Rahel 1996). In addition, natural or constructed barriers must be employed to prevent recolonization of restored reaches (Stuber et al. 1988, Rinne and Turner 1991, Pister 1998, Harig et al. 2000). Extensive monitoring over many years to confirm the recovery of the native species, as well as the absence of competitors, is also an essential part of any restoration plan (Rinne and Turner 1991, U.S. Fish and Wildlife Service 1993, U.S. Fish and Wildlife Service 1995, U.S. Department of the Interior 1995, Pister 1998, Harig et al. 2000). The need for these techniques, as well as the economic and political ramifications involved, precludes their use on larger rivers. Cherry Creek therefore represents an ideal compromise between treatable size and size needed to for a large restored population in a diverse area.

Results from more than three decades of restoration are mixed (Rinne and Turner 1991). Instances of success, such as the eradication of an introduced, and potentially catastrophic, population of brook trout from Arnica Creek in Yellowstone National Park (Gresswell 1991) are tempered by many failures (Rinne and Turner 1991, Harig et al. 2000). In fact, reintroduction programs are often expressed as a percentage of the instances where successful (stable and reproducing) populations have been established in relation to the total number of attempts to recover the species (Stuber et al. 1988, Rinne and Turner 1991, U.S. Fish and Wildlife Service 1995, Pister 1998, Harig et al. 2000). The percentages of successful restorations typically range from less than 40 % (Harig et al. 2000), to greater than 65 % (Rinne and Turner 1991). According to Rinne and Turner (1991), most restorations failed due to incomplete kill of the target species or unauthorized reintroduction. Examples of incomplete eradication are widespread and range from problems with electrofishing removal (Moore et al. 1983, Thompson and Rahel 1996, Shepard and Spoon 2000), to the use of unsuitable piscicides (Pister 1998). Pister's summary of the Kern River Golden trout recovery efforts points out another potential obstacle to complete eradication of non-natives that is shared with Cherry Creek. This is the need to apply toxicant to connected marshy areas to ensure eradication of young-of-the-year fish. Barrier structure failure is a common source of failure for many restoration projects, and is usually a result of improper design or materials, or severe hydrologic events (Rinne and Turner 1991, Pister 1998, Harig et al. 2000).

Perhaps most disturbingly, unauthorized reintroduction has been suggested as the reason for finding non-natives in reaches where previous reintroduction efforts were found successful (Rinne and Turner 1991, Harig et al.2000). Regardless of previous success rates, further implementation of practical restoration projects is mandated by the current emphasis on native fish recovery plans.

The pre-restoration data collected in this study will facilitate monitoring and evaluation of the restored westslope cutthroat trout population. By comparing the future restored population to the present community of non-native trout, a more specific evaluation of the degree of success of the restoration effort should be possible. Success of the restoration, and suitability of the site selected, could be evaluated by comparing the restored population's parameters to the capacity of the stream to support salmonids, as indicated by the parameters of the present non-native community. Such comparison would be possible with the demographic parameters of the non-native fish community determined by this study. Such pre-treatment data has generally been lacking in other restoration programs. In one instance where pre-treatment surveys were conducted, their purpose was to judge suitability of the site for reintroduction of Colorado River cutthroat trout (U.S. Department of the Interior 1995). More commonly, pre-treatment surveys were designed to determine distribution and abundance of the invasive, non-native species in order to decide where restoration efforts should be concentrated (Rinne and Turner 1991, U.S. Fish and Wildlife Service 1995, Pister 1998). Extensive before and after comparisons to estimated parameters of the present non-native community will allow for a more thorough evaluation of the restored native fish population, and perhaps may enable development of better recovery standards (Probst and Stefferud 1997).

Since the habitat found in Cherry Creek was judged suitable (Bramblett 1998), the parameters of the restored population such as biomass, density, condition, and growth rates should compare favorably to the baseline data taken on the existing non-native communities. This is in part due to similarity of the habitat requirements listed in the Habitat Suitability Index Models for the present species, rainbow trout (*O. mykiss*) and brook trout (*Salvelinus fontinalis*), and the restored species (westslope cutthroat trout) (U.S. Fish and Wildlife Service 1982a, 1982b, 1982c). Further support for this assumption is found in Platts and McHenry's (1988) summary of density and biomass of various trout assemblages in western streams, where no significant differences were found between densities and biomass in streams that had one trout species (allopatric), versus those that had more than one (sympatric).

The examples of similar densities and biomass for various stream salmonid assemblages suggest that elements of biological control, particularly competition, do not play a major part in determining overall productivity in most systems. Although some instances of non-native species removal have shown evidence of biological control, in the form of interactive segregation and niche shifts (Fausch and White 1981, Griffith 1988), there has been little evidence supporting resultant decreases in biomass due to less efficient resource partitioning (Nilsson 1967, Moore et al. 1983, Fausch 1988). Indeed, some studies showed inconclusive, or contradictory, evidence of niche shift, or displacement (Griffith 1972), or changes in biomass or standing crop (Moore et al. 1983), due to species removal. Ecological release is a common result in systems where a competitor is removed from a sympatric community with a depressed native population (Fausch and White 1981, Shepard and Spoon 2001). Shepard and Spoon (2001) found that higher biomass of the allopatric population versus the sympatric community was temporary, with the final result that the released population biomass levels returned to levels similar to those of the pre-restored sympatric community.

Conversely, the evidence for abiotic control, or physical factors, affecting trout densities and biomass is more convincing. Physical factors influencing density, biomass, and growth rates of trout populations include, but are not limited to: geomorphology (Platts 1979, Lanka et al. 1987); elevation (Lanka et al. 1987, Scarnecchia and Bergersen 1987, Probst and Stefferud 1997); percent suitable substrate (Lanka et al. 1987); gradient (Lanka et al. 1987, Probst and Stefferud 1997, Horan et al. 2000); percentage of undercut banks (Horan et al. 2000); width:depth ratio (Binns and Eiserman 1979, Scarnecchia and Bergersen 1987); flow (Herger et al. 1996); and pool quality (Lewis 1969).

The widespread similarities in demographic parameters such as biomass between sympatric and allopatric salmonid populations validate the use of the non-native, sympatric, salmonid parameters presented in this study for judging the relative success of the future Cherry Creek restoration effort. Comparisons of the community parameters in various sections of Cherry Creek that differ in physical habitat attributes may help elucidate which abiotic (habitat) factors contribute to the observed characteristics of the non-native salmonid community. By describing the existing non-native salmonid community and its relationship to the habitat, this study should provide a framework in which to judge not only the relative success of the future restoration, but may also help to determine if the restored westslope cutthroat trout population differs in its relationship with the physical habitat of lower Cherry Creek.

Study Purpose and Objectives

Purpose

The purpose of this study is to evaluate present salmonid populations and available habitat in three sections of lower Cherry Creek to provide baseline data for comparisons to future restored westslope cutthroat trout, and possibly Arctic grayling, populations. The comparisons can then be used to judge the relative success of the restoration as well as to help determine factors related to the observed characteristics of the restored westslope cutthroat trout population.

Objective 1

Estimate present fish demographic parameters and population characteristics in representative stream sections within lower Cherry Creek. Demographic parameters include population, density, and biomass estimates. Population characteristics include percent-at-age, mean length-at-age, growth rate, condition factor, and mortality estimates.

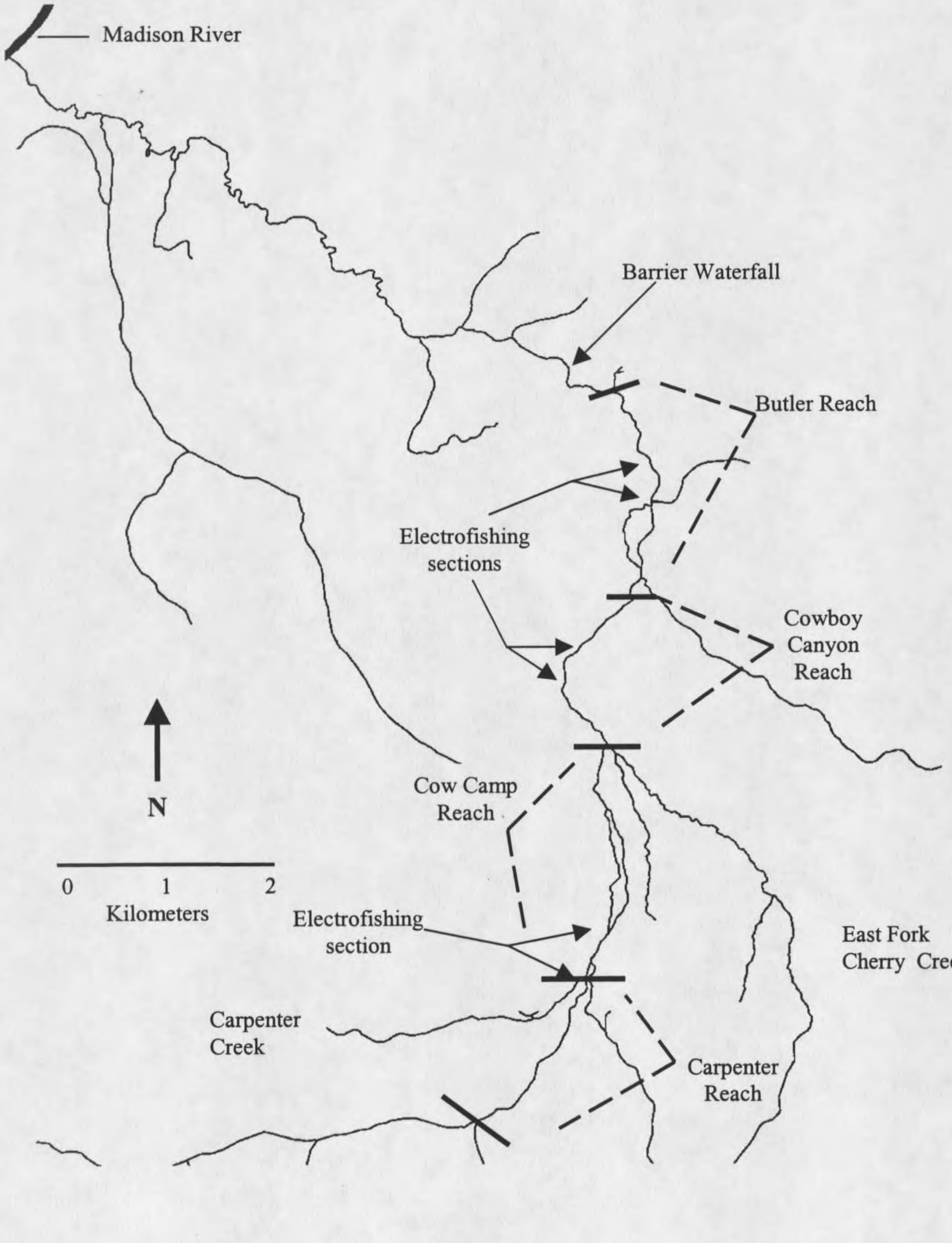
Objective 2

Compare estimated demographic parameters and population characteristics between reaches and species to determine whether significant differences exist and whether measured stream habitat features correlate with estimates of fish demographic parameters.

STUDY AREA

Cherry Creek flows in a northerly direction from the Spanish Peaks to the lower Madison River. From Cherry Lake, and other headwater sources, downstream to the barrier falls, the Cherry Creek drainage includes over 96 kilometers (60 miles) of streams judged to be suitable habitat for westslope cutthroat trout (Bramblett 1998). Several features of the Cherry Creek drainage (Figure 1) make it an ideal site for westslope cutthroat trout restoration. The drainage is blocked to any upstream non-native recolonization by a 7 meter (25 foot) high waterfall located approximately 10.5 kilometers (8 miles) upstream from its confluence with the Madison River. The waterfall may be a result of the geological history of the area, in that a major fault, coincidentally named the Cherry Creek Fault, underlies the area. The fault also delineates a major change in the parent material of the drainage, with limestone outcroppings becoming more common on the south, or upstream side of the fault (S. Custer, Dept. of Earth Sciences, Montana State University, personal communication 2001). The presence of limestone in the drainage may help explain the relatively high productivity of the stream, in terms of numbers of fish, as it is widely recognized that macronutrients dissolved from limestone increase the productivity of aquatic ecosystems (Platts 1979, Scarnecchia and Bergsen 1987). Cherry Creek's headwaters drain pristine National Forest lands, including a portion of the Spanish Peaks Unit of the Lee Metcalf Wilderness Area. The Flying D Ranch-Turner Enterprises Inc owns the lower drainage. This ownership has resulted in a comparatively unimpacted drainage, and simplifies the multi-jurisdictional

Figure 1. Map of lower Cherry Creek from mouth to top of Carpenter reach showing habitat inventory reaches and electrofishing sections.



cooperation required for a project of this scale. A qualitative habitat survey performed on the Flying D Ranch sections of Cherry Creek judged water quality, sediment levels, temperature, spawning and pool habitat, and riparian conditions as suitable for westslope cutthroat trout (Bramblett 1998). Presently, this habitat is inhabited by an abundant community of brook and rainbow trout.

METHODS

Study Reaches

Because the lower gradient areas of Cherry Creek were considered to contain the best habitat of the drainage available for restoration of westslope cutthroat trout and Arctic grayling (Kaya 1992), and because upper reaches were to be sampled by others, the selection of three contrasting reaches was limited to the portion of Cherry Creek between the barrier waterfall and the higher gradient headwaters located above Cow Camp basin. Two of the three principal reaches, Butler and Cow Camp, were chosen based on their lower gradient and abundance of high quality pool habitat. The upper reach (Cow Camp) was lower order, third versus fourth, and had a slightly higher gradient, 1.5 versus 1.0 percent observed gradient, than the Butler reach. The third principal reach (Cowboy Canyon) was chosen as a contrast and to establish the amount of fish that this middle, and sometimes intermittent, reach could support. In addition, a fourth, transitional reach (Carpenter Reach) was surveyed for habitat characteristics so that all lower areas of Cherry Creek were surveyed similarly. Although this reach contained a small amount of lower gradient habitat resembling the Cow Camp reach, it was predominantly comprised of higher gradient habitat more associated with the headwater areas of Cherry Creek. Due to this, and because a majority of the reach was upstream of the study area (from the barrier falls upstream to Cow Camp Basin), the

Carpenter reach did not contain an electrofishing section; and is only included to provide an example of the habitat found upstream of the study area.

Survey reaches were numbered as follows: the Butler reach as reach number 4, Cowboy Canyon as reach number 5, Cow Camp as reach number 6, and finally reach number 7 (Carpenter reach). Reach numbering began at four in order to accommodate future numbering of reaches below the barrier waterfall.

Macrohabitat Parameters

The three principal study reaches for this project were located in the lower portion of the drainage above the barrier falls (Figure 1). The Butler Reach began just upstream of the barrier falls and continued for approximately 2.4 kilometers (1.5 miles) upstream to a barrier diversion dam. A meandering channel ("C type" after Rosgen 1994), deep meander pools, and an abundance of willow (*Salix spp.*) cover characterized this reach. A diversion dam marks the downstream end of the next reach known as Cowboy Canyon. This reach, as its name implies, was more confined, yet was still low gradient (1.2 %). It encompassed approximately 2.8 kilometers (1.7 miles) of stream between the dam and the mouth of East Fork Cherry Creek. Though reported to become intermittent in areas during drier years (C. Francis, former Turner Enterprises employee, personal communication), flows were maintained through this reach during the two years of this study. The third principal reach began at the mouth of the East Fork and continued 4.2 kilometers (2.6 miles) upstream to the mouth of Carpenter Creek. Here the valley bottom opened up again into an area known as "Cow Camp". The stream here resembled a slightly smaller and higher gradient version of the Butler Reach. The Cow Camp section also had numerous beaver (*Castor canadensis*) dammed backwater areas off the main channel.

To quantify and qualify stream habitat, macrohabitat measurements were taken in each reach in accordance with USDA Forest Service's R1/R4 (Northern/Intermountain Regions) fish habitat inventory procedures (Overton et al. 1997). Measurements included: habitat type (pool type or fast-water type), length of habitat unit (HU), HU width, HU average depth, maximum and crest depths (pool HU), number and average maximum depth of pocket pools (fast water HU), percent surface fines and substrate composition (both visually estimated and measured by Wolman (1954) pebble counts and surface fines grate measurements), bank stability (percent estimate), number of large woody debris pieces (including singles, aggregates and root wads), and dominant and subdominant riparian vegetation types. Measurements of length and width were performed using a tenth-of-meter measuring tape or hip chain, while depths were measured using a tenth-of-meter stadia rod.

Percent channel gradient and water temperature measurements were also taken during surveys. Percent gradient was estimated by dividing stadia rod-measured, vertical rise by hip-chain horizontal distance measurements (tenth-of-meters). Photographs were taken and comments were recorded for unusual features, reach breaks, etc. As recommended by Overton et al. (1997), the habitat inventory was performed at base flows during mid-summer to enable proper habitat type identification and future comparisons.

Habitat data was entered into FBASE (USDA Forest Service 1998), a stream habitat data base program, for analysis. This software generated reports that analyzed habitat parameters by reach, such as habitat area (total or mean for the survey reach), volume, mean maximum pool depth, etc. (Wollrab 1998). Habitat measurements were archived on disk and are available through the Ecology Department, 304 Lewis Hall.

This habitat inventory procedure used can be performed at various levels of intensity according to habitat parameters recorded. For this study habitat was surveyed at inventory level II (Overton et al. 1997). This level includes the basic habitat measurements of level I such as habitat length, width, average and maximum depths, etc., and includes further macrohabitat measurements such as: substrate composition, side channel measurements, riparian vegetation types, and large woody debris (LWD) counts. Level II differs from the most intense level of inventory, level III, by not including measurements of bank lengths and LWD dimensions. The level II method was employed primarily because the LWD measurements required for the level III method are more applicable to systems where willow is not the dominant source of LWD (Overton et al. 1997).

Fish Demographic Parameters

Population Estimates

Population estimates were conducted on monitoring sections within each of the three principal reaches in late summer, August 24-28, 1998 and 1999, except for the Butler reach for 1999 which, due to scheduling with Turner Enterprises, was delayed until September 17, 1999. Locations and lengths of each monitoring section were chosen to encompass representative habitat and their approximate frequencies for each respective reach. Estimates were conducted within block-netted monitoring sections of at least 400 m in length, from 402.6 m (Cow Camp), to 492.6 m (Butler), to 512.3 m (Cowboy Canyon). Locations of the sections were confirmed with GPS measurements: 45^0 33'55" N by 111^0 26'66" W for the bottom end and 45^0 33'43" N by 111^0 26'61" W for the top end of the Butler section, 45^0 32'66" N by 111^0 27'28" W and 45^0 32'46" N by 111^0 27'28" W for the top and bottom of the Cowboy Canyon section, and 45^0 30'55" N by 111^0 26'98" W and 45^0 30'38" N by 111^0 27'02" W for the top and bottom ends of the Cow Camp section. Sections varied in length to encompass representative habitat units, and to avoid having electrofishing section boundaries occurring in the middle of habitat units. Length of the sections also helped ensure that extrapolations of the estimates to a longer common unit of length (km) were more accurate than would have been the case if shorter sections were used.

Fish were captured using mobile electrofishing. The electrofishing setup consisted of a gas-powered generator that supplied electricity through a "Leach Box" rectifier (Dr. Harvey Leach, Department of Electrical Engineering, Montana State University) set for an output of 300-400 volts of direct current (DC). This current was applied through the use of a mobile anode; the cathode was attached to an 11-foot-long Coleman "Crawdad" boat that carried the generator, rectifier and a large tub for holding fish. This setup provided the power necessary to sample the larger volume pools found in the Butler reach. Although cumbersome in smaller, upstream sections, this equipment enabled more efficient depletions due to the power provided, the ability to shock continuously for multiple hours, and the range and versatility of a mobile anode. Additionally, because large numbers of fish were caught, the ability to transport these captured fish in a large tub located in the boat meant that for more time could be spent electrofishing, instead of shuttling captured fish to holding pens. Estimates were obtained using a two, or three-pass depletion technique (Zippin 1958, Seber and Le Cren 1967). Field assessments of overall efficiency (capture probabilities greater than 70 %) resulted in all but the Butler section in 1999 conducted as two-pass depletion estimates.

Program Micro-Fish (Van Deventer and Platts 1985) was used to provide population estimates, capture probabilities, and their standard errors. The assumption of geographic closure of the sampled population was met by the use of block nets. Equal effort expended on each pass, and the omission of estimates for young-of-the-year size fish helped to meet the assumptions of equal effort and capture probabilities (Seber and Le Cren 1967, White et al. 1982). As will be seen in the results, population estimates excluded age-0 size fish. This is because capture of young-of-the-year fish in stream electrofishing is generally very inefficient (Lagler 1956, Raleigh and Short 1981, Riley and Fausch 1992, Reynolds 1996). Because the section length varied, population estimates were also expressed as number of fish per km. Calibrating these estimates to a common unit of length enabled an easier comparison with other studies.

Density and Biomass Estimates

Density estimates (number of fish per m^2) were obtained by dividing population estimates for each species, section, and year by the total surface area of the associated section. Surface area was determined by summing the surface areas for all individual habitat units in the respective shocking section.

Biomass estimates (grams per m^2) were estimated by calculating average weights for each species, section, and year, multiplying these values by the respective population estimates, and dividing the resulting products by the total surface area of the respective section. Lengths and weights of fish were measured using a tenths-of-foot measuring board and a tenths-of-pound spring scale; units were later converted to metric (millimeters and grams). Each species' contribution to combined biomass was also recorded as percent of total biomass.

Values of fish or grams per m^2 enable direct comparison to like units used by Platts and McHenry (1988), as well as to other studies where like units were used (Horan et al. 2000, Hilderbrand and Kershner 2000, Shepard and Spoon 2001). Both density and biomass estimates excluded young-of-the-year fish because they were not well represented in the population estimates due to limited sampling efficiency for this size class (Riley and Fausch 1992, Reynolds 1996).

Length-Frequency Distributions

Length-frequency distributions were plotted for each species in each section by year, using a 25.4 mm (1 inch) size interval. Length-frequency histograms were used to infer age structure and mortality. Peaks in histograms were assumed to represent the most common length for a particular age in a population. Analysis of relative strength of each year class was limited to age 1 and older fish because efficiency of capture for young-of-the-year fish was very low (Lagler 1956, Raleigh and Short 1981, Reynolds 1996). Although age-0 fish were under-represented in the length-frequency histograms, the capture of some age-0 sized-fish in most sections enabled me to approximate their length distributions. Ages inferred from length-frequency distributions were also compared with ages assigned from scale and otolith readings (Mackay et al. 1990).

Fish Population Characteristics

Age

While age structure was determined by investigating length-frequency distributions, ages of individual fish were assigned using scales. Scales were collected from a subset of fish of representative lengths during the depletion estimates. The large numbers of fish precluded the aging of each individual fish; accordingly, scales were taken from approximately 10 fish of representative size classes for each species in each section to facilitate construction of an age-length key, described below (Devries and Frie 1996). Scales were scraped from the sides of the fish in an area just posterior to the dorsal fin and just above the lateral line (Mackay et al. 1990). The scales were then placed in sample envelopes on which pertinent information was recorded such as, species, length, weight, section, and date. Scale samples were then pressed into acetate sheets to facilitate reading (Wayne Black, Montana Fish, Wildlife and Parks). Unfortunately, many of the scale samples from many of the older, longer fish were comprised of only regenerated scales that could not be aged.

Otoliths were removed from some of the longer size-classes of fish during the second season for scale verification purposes. By comparing the age of fish determined by scale readings, to the otolith readings from the same fish, evaluation of the accuracy of scale readings is possible (Mackay et al. 1990 and Devries and Frie 1996). Otoliths were removed from fish by cutting away the dorsal portion of the head to reveal the brain case, after which the otoliths were carefully extracted and placed into envelopes labeled as described above for scales. Otoliths were mounted on slides and ground to their center axis to facilitate accurate readings. The prepared otoliths were then back-lighted under a dissecting microscope and the darker annuli were counted and recorded (Brothers 1987).

First year annuli are often missing in inter-mountain salmonid populations, particularly slower growing ones (Lentsch and Griffith 1987). Comparisons of scale readings to length-frequency distributions, back-calculated mean length-at-age, and otoliths demonstrated that missing annuli were not present for fish sampled in this study. A smaller number of otoliths were read for verification of ages determined from scales. Otoliths were needed for verification because scales become less reliable for aging trout over three years in age. Otoliths provided verification for older fish as well as for aging brook trout, which with their much smaller scales with crowded annuli (growth rings), can be difficult to age from scale readings alone (Mackay et al. 1990, Devries and Frie 1996). Ages interpreted from scales and otoliths were verified by having another person experienced with scale and otolith-aging techniques review my results.

Percent-at-Age and Mean Length-at-Age

Calculating the percent-at-age and mean length-at-age of captured fish necessitated estimating ages for all size classes of fish captured. After determining ages for a subset of fish within various size classes, ages could then be assigned to all fish to determine percent-at-age and mean length-at-age (Devries and Frie 1996). The construction of such an age-length key allows for assigning ages to un-aged fish in the same size class. Of particular importance is that size class subsets that have more than one age are used to assigned ages according to the ratio of the subset of aged fish in that size class. However, due to some scale samples being comprised of regenerated scales, many of the sample sizes used for assigning age ratios for some size classes were smaller (less than five) than anticipated.

Due to the inefficiencies of capture for the age-0 size-class of fish, percent-at-age values were limited to age-1 and older fish. Although this omission failed to describe the contribution of age-0 fish to the community, it still allowed for comparisons of the relative contributions of the older ages to the population. In sections where age-0 sized fish were captured, mean length-at-age values were calculated for all ages.

Annual Growth Estimates

Annual growth increments were estimated through back-calculation for collected scales. In addition, annual growth increments were also determined from differences in mean length-at-age of fish captured from consecutive age classes over the two years. Because this latter method depended on differences in mean lengths-at-age of captured fish over consecutive years, it only provided one year of annual growth estimates. Comparisons between the two methods helped in elucidating possible biases in the back-calculated estimates (Devries and Frie 1996). Back-calculation was performed with scales, while otoliths were used as a means of age verification.

Although Weisberg back-calculation analysis is generally preferred, it requires larger sample sizes and samples from multiple years, which were not available in this study; therefore, the less rigorous Fraser-Lee method was employed (Devries and Frie 1996). Under this method the distance of verified scale annuli from the scale's focus were measured along with the distance of the scale's margin from the focus. Fish lengths at previous annuli were estimated according to the Fraser-Lee formula:

$$Li = \frac{Lc - a}{Sc} Si + a$$

where

$$\frac{Lc - a}{Sc} =$$

the slope of a two-point regression line to estimate Li, a = intercept parameter and Li, Lc,Sc and Si are defined as:

Li = back-calculated length of the fish when the ith increment was formed,

Lc = length at capture,

Sc = radius of hard part at capture, and

Si = radius of the hard part at the ith increment.

The resultant back-calculated lengths-at-age were averaged for each age class. Average annual growth was estimated by subtracting back-calculated average lengths at two age classes (Carlander 1969, Mackay et al. 1990, Devries and Frie 1996). For example, growth increments between back-calculated mean length-at-age 1 and back-calculated mean length-at-age 2 was considered age 1 growth (Carlander 1969). Because there were no previous means lengths-at-annuli to subtract, and use of the intercept parameter was problematic due to its dependant nature on regressions of various power, annual growth increments for age-0 fish were not included in the back-calculated estimates. However, using differences between mean lengths-at-age data for fish of consecutive age classes over consecutive years did allow me to approximate annual growth for some age-0 fish. For example, although under-represented, an estimate of mean length-at-age 0 was obtained for some species in some sections in 1998. When this length was subtracted from mean length-at-age estimates for age-1 fish of the same species and section in 1999, an approximation of annual growth for age-0 fish was obtained.

The back-calculated mean lengths-at-age represented lengths at time of annulus formation. Consequently, these values were smaller than mean lengths-at-age values reported for fish captured later in the season. Because annuli were formed in the period immediately prior to the growing season, differences between the two techniques for estimating mean lengths-at-age were a result of growth since annulus formation (Devries and Frie 1996). Additionally, reported back-calculated increments for age classes not followed by an additional age (annuli) were pessimistic as the back-calculated length in these instances represents the distance between last annuli and scale margin and was therefore not representative of an entire years' growth (Carlander 1969).

Condition Factor Estimates

Fulton's condition factor K (Devries and Frie 1996), was calculated for all fish; however, due to the inaccuracy of spring scale readings, only fish 100 mm or longer were used in comparisons for differences in K. The scale inaccuracy was limited to small fish and their associated small weights. This resulted in the small, less than 100 mm long, fish having weights and K values biased high. The use of K values for fish greater than 100 mm in length, typically age 1 and above, enabled better comparisons to be made between species, sections, years, and with other studies where K was used.

Fulton condition factor K, was calculated for each fish following the formula:

$$K = (W/L^3) * 100,000$$

Where W was weight in grams, and L was total length in millimeters. The resultant product is an index of the overall condition of the fish. Values for fish of average condition are typically around 1.0, with values less than this indicating poor condition fish and values much over 1.0 suggesting fish of better than average condition (Carlander 1969, Anderson and Neumann 1996).

Mortality Estimates

Mortality was estimated by establishing age composition of the sampled populations. Estimation of populations and mean lengths-at-age enabled calculation of annual survival for the three monitoring sections. Mortality, the inverse of survival, was estimated separately for each species, section, and year. With the few age classes present, and the collection of only two years' of data, annual survival was calculated using the simplified formula based on coded age after Robson and Chapman (1961). In this method annual survival (S) is estimated by the following formula:

$$S = \frac{T}{n + T - 1}$$

Where n is the total number of fish, beginning with the first age class accurately estimated, and T is determined from the summation of coded age totals and their respective coefficients. The 95% confidence interval, assuming normal distribution of error was calculated by doubling the square root of the estimate of variance (s). Variance was calculated using the formula:

$$s = \frac{T}{n+T-1}\left(\frac{T}{n+T-1} - \frac{T-1}{n+T-2}\right)$$

Statistical Analysis

Comparisons of parameter means were accomplished using one-way ANOVA from the computer program Minitab (1998). Data analyzed were tested for normality by referring to histograms and normality plots of residuals. Comparisons of means of fish demographic parameter estimates such as, population per common unit length, density, and biomass required the use of the two-year's estimates as replicates. In these instances, the associated low degree of freedom (2) meant that only large differences were found significant. Comparisons of fish population characteristics such as, mean length-at-age, annual growth, and condition factor were obtained from the means of many individual fish measurements and as a result were much more robust.

Tukey's pairwise comparison procedure was used in the single factor one-way ANOVA analysis. In a few instances, analysis of only two population means was performed using a two-sample T-test. A significance level of $\alpha = 0.05$ was used for all tests.

Stepwise regression was used to determine which predictors (habitat parameters) best described the response variable (fish population characteristics and demographic parameters). After the best subset of predictors was chosen, each correlation was obtained through the use of a single, simple linear regression. Of the correlations found to be significant ($\alpha = 0.05$), percent variability explained (R-squared values) was recorded. As with the ANOVA analysis, regressions were performed using the statistical software Minitab. Simple linear regressions were also used for correlating length-scale relationships.

RESULTS

Macrohabitat Parameters

Habitat Summary

F-Base summary reports are presented in Appendix A; however, a brief review of the habitat inventory results is summarized below (Tables 1 and 2). As the reaches progress upstream, and Cherry Creek becomes smaller, some expected trends become evident. Slow-water (pool) habitat types consistently decrease in both mean maximum depth, mean width, and mean length. Mean maximum depth for pools decreases from 1.04 m in Reach 4 (Butler) to depths of 0.95 m and 0.94 m in Reach 5 (Cowboy Canyon) and Reach 6 (Cow Camp), respectively. Mean channel width also decreases throughout the reaches, from 7.9 m in the Butler reach (reach 4), to 7.2 and 6.8 m for the respective upstream reaches. Other trends evident, except for a slight departure in the Cow Camp reach (reach 6), is that the characteristic percent of slow water habitat type by length, i.e. (pool:fast-water ratio) decreases progressively in an upstream direction, from 32.4 % (approximately 1:3) in the Butler reach, to 19.8 % (approximately 1:5), 23.4 % (approximately 1:4) and 12.1 % (approximately 1:8) for the Cowboy Canyon, Cow Camp, and Carpenter reaches respectively (Table 1 and Figure 2). The much lower value for the Carpenter reach is indicative of its lower habitat potential and confirms that the three principal reaches represent the majority of the quality habitat found in lower Cherry Creek. Cow Camp's departure from this trend suggests that this reach is second only to the Butler reach in terms of providing quality habitat in the form of greater amount of pool habitat.

Table 1. Summary of main channel habitat dimensions (m), by reach and habitat class.

Reach: 4 (Butler) Type: C

Habitat Class	Habitat Length Total	Habitat Length Mean	Percent	Mean Width	Habitat Depth Mean	Habitat Depth Max	Width: Depth
Riffle	1,000.2	38.5	42.4	8.4	0.24		37.5
Run	594.6	37.2	25.2	7.2	0.33		22.4
Pool	762.8	21.2	32.4	7.7	0.43	1.07	18.4
Totals	2,357.6	30.2		7.9	0.32		27.5

Reach: 5 (Cowboy Canyon) Type: C

Habitat Class	Habitat Length Total	Habitat Length Mean	Percent	Mean Width	Habitat Depth Mean	Habitat Depth Max	Width: Depth
Riffle	1,525.0	35.5	55.1	7.0	0.20		37.2
Run	695.6	30.2	25.1	7.8	0.25		32.5
Pool	546.8	16.1	19.8	6.8	0.38	0.95	18.5
Totals	2,767.4	27.7		7.2	0.25		32.3

Reach: 6 (Cow Camp) Type: C

Habitat Class	Habitat Length Total	Habitat Length Mean	Percent	Mean Width	Habitat Depth Mean	Habitat Depth Max	Width: Depth
Riffle	2,426.0	39.8	57.6	7.1	0.21		35.5
Run	796.4	22.8	19.0	6.1	0.28		22.2
Pool	986.3	15.9	23.4	6.5	0.37	0.94	18.1
Totals	4,208.7	26.8		6.8	0.26		29.0

Reach: 7 (Carpenter) Type: B

Habitat Class	Habitat Length Total	Habitat Length Mean	Percent	Mean Width	Habitat Depth Mean	Habitat Depth Max	Width: Depth
Riffle	3,003.1	75.7	80.9	6.3	0.18		37.3
Run	259.8	20.6	7.0	4.9	0.24		21.0
Pool	449.5	12.5	12.1	6.0	0.34	0.88	19.1
Totals	3,712.4	41.2		6.2	0.20		34.0