Minimum stream length requirements for McCloud River redband trout (*Oncorhynchus mykiss spp*) in Trout and Tate creeks, Siskiyou County, California

by

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ABSTRACT

Minimum Stream Length Requirements for McCloud River redband trout (*Oncorhynchus mykiss spp*) in Trout and Tate creeks, Siskiyou County, California

I located existing and potential barriers to fish movement, evaluated habitat, and estimated minimum stream lengths required to maintain genetically viable populations of McCloud River redband trout (*Oncorhynchus mykiss spp*) in two streams, Trout and Tate creeks, in the upper McCloud River basin in northern California. The goal of this research was to evaluate the potential effectiveness of isolation management to protect and restore populations of redband ttout. Minimum stream length requirements were based on reach-scale estimates of fish density and survival. Population estimates were obtained using a modification of the Hankin and Reeves (1988) approach and from spotlight surveys. Redband trout were the only salmonid species observed in Trout Creek while Tate Creek contained redband trout and brook trout (*Salvelinus fontinalis*). Redband trout densities in summer 2009 were approximately 378 km^{-1} and 652 km^{-1} in Trout and Tate creeks, respectively. Estimates of minimum stream length needed to maintain a population of 2,500 individuals with densities observed in 2009 ranged from 7.35 to 13.23 km on Trout Creek and from 4.26 to 7.67 km on Tate Creek. Abundance differences in streams were probably due to availability of warm, retentive, high productivity habitat in the lower reaches of the larger Tate Creek watershed. Trout Creek was determined to be a poor candidate for deliberate isolation because a percolation barrier isolates it from the upper McCloud River and further barriers would fragment already limited habitat. Although trout from the lower reach of Tate Creek showed

morphological evidence of hybridization, the stream supported a higher trout density and available habitat exceeded estimated minimum stream length. As such, it may represent a viable isolation candidate, with sufficient resources to support growth of translocated populations with greater genetic purity. Existing culverts on both streams do not appear to significantly fragment habitat.

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INTRODUCTION

McCloud River redband trout are a distinctive form of rainbow trout
(*Oncorhynchus mykiss*) indigenous to the Upper McCloud watershed. They are part of a
complex of Northern Sacramento River redband trout, which also includes trout from the
Pit and Feather rivers and from the headwaters of the Sacramento River (Behnke 2002).
McCloud River redband trout are characterized by a brick-red band along the lateral line,
cutthroat-like slash marks, irregular parr marks, and orange-tipped dorsal, anal, and
ventral fins. They have several physical characteristics, such as basibranchial teeth and
yellowish undertones laterally, thought to be ancestral to *Oncorhynchus mykiss* (Nielsen
et al. 1999).

The phylogenetic position of McCloud River redband trout within *Oncorhynchus
mykiss* is uncertain. The fish is believed to be a relict subspecies of non-anadromous
rainbow trout of ancient coastal origin which has adapted to harsh, fragmented
environmental conditions (Behnke 2002). It has a close genetic association with Baja
Mexico trout (*O. mykiss nelsonii*) and likely had a southern source population that
migrated north with a warming climate (Nielsen et al. 1999). The Mt. Shasta area may
have offered a zone of refuge during the Pleistocene glaciation (Behnke 2002) and
continued isolation via volcanism could have maintained redband ancestral traits (Nielsen
et al. 1999). Development of the Upper and Middle Falls on the mainstem McCloud
River later isolated trout in the Upper McCloud watershed from the Sacramento River
basin. The Sheepheaven Creek redband trout population is believed to best represent the
earliest colonizer of the northern Sacramento system (Behnke 2002).

Identification of original redband trout distribution boundaries is complicated by genetic introgression from hatchery rainbow trout and translocation of Sheepheaven Creek redband trout to other upper McCloud tributaries. All redband trout in upper McCloud streams surveyed by Simmons et al. (2010) contained Sheepheaven redband trout alleles. This suggests historic gene flow between populations prior to hatchery introgression events and calls into question Behnke's (2002) contention that Sheepheaven Creek fish warrant separate subspecies designation. Putative redband trout populations currently exist in Trout, Tate, Sheepheaven, Edson, Swamp, Moosehead, Bull, Dry, Blue Heron, Shady Gulch, and Raccoon creeks as well as the mainstem McCloud above Middle Falls. Although the exact subspecies status and genetic integrity of McCloud River redband trout are still under question, the fish are highly valued for their unique appearance and ability to persist in harsh and isolated environments of limited habitat.

Redband trout are currently threatened with displacement by nonnative brook (*Salvelinus fontinalis*) and brown (*Salmo trutta*) trout and genetic introgression from introduced rainbow trout hatchery stock. Nonnative salmonids are present in much of the McCloud River watershed (United States Department of Agriculture 2008 and Simmons et al. 2010). Invasive species can displace natives through predation and competition which can reduce individual growth rates (Heggenes 1988, Magoulick and Wilzbach 1998, Fausch et al. 2006, McHugh and Budy 2006, and McGrath and Lewis 2007). Introgression produces a reduction in phenotypic diversity and results in loss of divergent evolutionary history and adaptation to local environments (Allendorf and Leary 1988). The process of introgression in the Upper McCloud River probably began with stocking of hatchery rainbow trout in 1957 (United States Department of Agriculture 2008).

Remaining stocks of relatively non–introgressed redband trout are relegated to tributaries of the Upper McCloud River (Simmons et al. 2010 and Nielsen et al. 1999).

Because of threats posed by nonnative salmonids and harmful land use practices, McCloud River redband trout were listed as a Candidate species under the Federal Endangered Species Act (United States Department of Agriculture 2008) and as a California species of special concern (Moyle 2002). Through the efforts of private and government entities, a 1998 conservation agreement led to the eventual removal of redband trout from the Candidate species list. Despite de-listing, threats from introduced salmonids remain and potential methods to preserve redband trout are under evaluation (United States Department of Agriculture 2008).

One conservation strategy under consideration involves deliberately isolating remaining redband trout stocks above stream barriers. Failure to isolate native populations may result in extinction from the threats posed by invasion and hybridization (Kruse et al. 2000, Shepard et al. 2005, and Fausch et al. 2009). However, isolation to small, headwater reaches imposes an additional suite of risks. Barriers may prevent individuals from re-colonizing unpopulated reaches after local extinction. Migratory life history types and genetic exchange between populations are eliminated with isolation. Concentrated populations are vulnerable to extinction through stochastic events (Shepard et al. 2005 and Fausch et al 2006). Volcanism, drought, and catastrophic hillslope failure could eliminate small redband populations in the Upper McCloud River. Low-order streams are usually less productive than reaches lower in the watershed, limiting population size and individual growth rates (Harig et al. 2000). Isolated reaches may not provide area sufficient to support effective population size necessary for maintenance of

phenotypic diversity (Fausch et al. 2009). Effective population size is defined as the

number of individuals within a population which are successfully reproducing (Watters et

al. 2003). A minimum of 500 individuals is generally agreed upon as the amount needed

to sustain long term genetic viability. If the effective population makes up only 10-20%

of the overall population, 2,500 to 5,000 individuals would be required to produce an

effective population of 500 (Nelson and Soulé 1987).

For isolation management to be successful, deliberately isolated reaches must

provide protected stream length sufficient to maintain effective population size. Harig et

al. (2000) found that successful relocations of cutthroat trout required at least 5.7 km of

stream length in Colorado streams. Harig and Fausch (2002) found cutthroat trout

relocations were successful in watersheds of at least 15 km^2 which contained streams

about 5 km in length. Minimum habitat requirements for long-term viability probably

vary by species and stream productivity.

Hilderbrand and Kershner (2000) used fish abundance to estimate minimum

stream length (MSL) required for population viability as:

$$\text{MSL} = N(1 - s)^{-1} \cdot D^{-1}$$

where N is the number of individuals needed to produce a desired population by the next

spawning period, D is the number of fish per unit stream length, and s is the proportion of

fish exiting the population through mortality or emigration prior to the next spawning

event.

As mortality increases or fish abundance decreases, the length of stream necessary to arrive at the desired population increases. With 2,500 individuals as a target population size, they estimated that 8 km of stream were necessary to maintain a population with high abundance (0.3 fish/ km^2). Harig and Fausch (2002) identified several habitat parameters influencing successful translocation of greenback cutthroat trout in Colorado including reach slope, stream width, number of deep pools, and structure within pools. Quantification of habitat features among streams that vary in their capacity to support fish may reveal physical characteristics that play a crucial role in affecting productivity.

Immediate threats to genetic integrity from hatchery rainbow trout and displacement from non-native trout can outweigh long-term stochastic demographic and environmental risks introduced by isolation (Shepard et al. 2005). Protection of remnant populations and eventual reseeding of recovered habitat represent a viable short-term solution (Harig et al. 2000 and Shepard et al. 2005). However, deliberate isolation without prior knowledge of basic population parameters could result in extirpation of populations too small to maintain long term genetic viability. Investigation of redband trout population structure is therefore necessary to identify minimum habitat required for a successful isolation management strategy.

The objectives of this study were to: (1) describe and compare population structure (abundance, distribution, and survival), fish age, growth, and condition for redband trout in two tributaries in isolation from and connected to the mainstem McCloud River; (2) estimate minimum stream length required to support genetically viable populations in each stream; (3) determine effectiveness of existing barriers and

possible fragmentation of candidate isolation reaches; and (4) evaluate habitat features that may be associated with differences in minimum stream length between the two streams. This information has not previously been collected on a reach-wide scale in the upper McCloud basin.

Study Site Description

The upper McCloud River catchment drains 1,487 km^2 in northeastern California at the foot of Mount Shasta, a dormant volcano rising 4,316 meters above sea level (Figure 1). The upper river flows west for 39 kilometers from its origin at an elevation of 1,897 meters to Middle Falls at 885 meters elevation. The McCloud River drains to the Sacramento River and about 40 % of the catchment is privately owned. Annual precipitation, mainly in the form of winter snow, averages 114 centimeters per year (United States Department of Agriculture 2008). Higher tributary reaches are subject to freezing, primarily due to low flows in winter (personal communication, S. Bachmann 2009. U.S. Forest Service Hydrologist, Shasta-Trinity National Forest, 2019 Forest Road, McCloud, CA 96057).

Habitat in smaller tributaries is limited by steep gradients and low flows. Previous surveys summarized by the Forest Service (United States Department of Agriculture 2008) revealed "poor to fair" habitat conditions in small tributary streams with habitat types dominated by riffles and runs. The larger tributaries, including Tate and Trout creeks, have been described as containing sufficient spawning gravels, a "good mix" of riffles and deep pools (greater than one meter in depth), "good" cover, and an abundance of habitat associated with large woody debris (United States Department of Agriculture 2008).
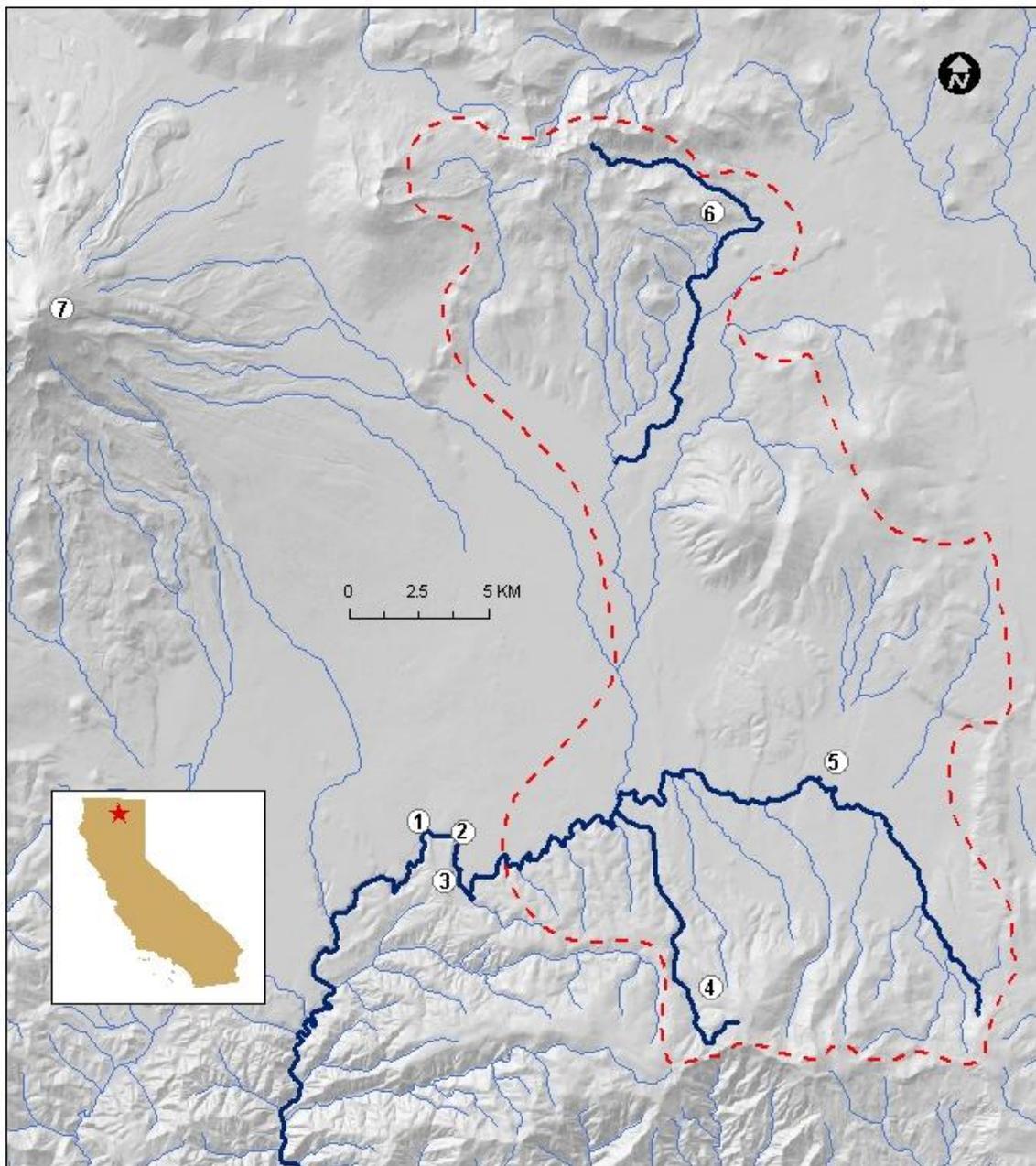
Figure 1. Overview of the upper McCloud River basin in northern California, study streams, and redband trout refugium boundary. The refugium boundary is depicted with a dashed red line. Features identified include: (1) Fowler's campground; (2) Middle Falls; (3) Lakin Dam; (4) Tate Creek; (5) Mainstem McCloud River; (6) Trout Creek; and (7) Mount Shasta

The upper McCloud basin has an extensive history of timber harvest activity and livestock grazing. Commercial logging began in the late 1800's and was not highly regulated until 1973. Harvest activity is currently overseen by state and federal agencies. Private landowners voluntarily participate in watershed protection activities such as culvert replacement and road removal (United States Department of Agriculture 2008). Grazing use in the upper McCloud basin began in the 1800's and reached its peak in the 1940's. Today only Trout Creek remains inside a grazing allotment and its riparian zone in lower reaches has been protected by fencing (United States Department of Agriculture 2008).

Because of concerns of the California Department of Fish and Game (CDFG) about sampling a species of concern in limited habitat, a collection permit was obtained only for Trout and Tate creeks in the upper McCloud basin. Both are located within the redband trout refugium (Figure 1) and are subject to sport fishing. Current regulations allow a bag limit of five salmonids per day.

Trout Creek is a second-order stream draining approximately 1,154 hectares upstream of Trout Creek campground. Trout Creek enters the mainstem McCloud basin from the north where lava flows and fluvial deposits originating from Mt. Shasta and Medicine Lake volcanoes have created porous volcanic soils that promote significant percolation of surface waters (Selters and Zanger 1989, Nielsen et al. 1999, and United States Department of Agriculture 2008). Many of the McCloud's northern tributaries, including Trout Creek (Figure 1), experience subsurface flow even in normal runoff years when they contact this porous substrate with resulting isolation from the mainstem (Nielsen et al. 1999 and United States Department of Agriculture 2008). The stream

length of suitable habitat in Trout Creek can range from 6.1 km in a normal runoff year to 5.6 km during dry years (United States Department of Agriculture 2008). Temperature in Trout Creek ranged from 5 to 8.5 ^{o}C during summer surveys (personal observation). Upper Trout Creek contains only redband trout while brown trout have been captured in the reach downstream of Trout Creek campground (personal communication, S. Bachmann 2009. U.S. Forest Service Hydrologist, Shasta-Trinity National Forest, 2019 Forest Road, McCloud, CA 96057). Trout Creek was stocked with hatchery rainbow trout prior to chemical treatment in 1977. Rainbow trout from Sheepheaven Creek were translocated to Trout Creek post treatment and hatchery rainbow have not been introduced since (United States Department of Agriculture 2008). Fish in Trout Creek display a close genetic relationship to Sheepheaven redband trout (Nielsen 1999). However, Simmons (2010) discovered evidence of hatchery rainbow trout lineage in Trout Creek redband, suggesting hatchery fish survival of chemical treatment and subsequent introgression.

Tate Creek is a second-order stream draining approximately 1,503 hectares. Tate Creek, and other upper watershed tributaries entering from the south, flow through tertiary lava flows and account for the majority of surface runoff to the main stem. Tate Creek contains as much as 10 kilometers of suitable habitat in normal runoff years and as little as 8 kilometers in dry years. Soils here consist of ashy, sandy loams and support mixed conifer forests (United States Department of Agriculture 2008). Temperatures recorded during summer surveys ranged from 8 to 12.5 ^{o}C. Hatchery trout are currently stocked in the mainstem McCloud River downstream of the Tate Creek confluence near Fowlers Campground and in McCloud Reservoir. Stocking above Middle Falls was

terminated in 1994, after which recreational use decreased dramatically. Lakin Dam

(Figure 1) is actively stocked with brook trout (Nielsen 1999, and Simmons et al. 2010).

The fish assemblage in Tate Creek includes: redband trout, coastal rainbow trout

(*Oncorhynchus mykiss spp*), brown trout, and brook trout (United States Department of

Agriculture 2008). Tate Creek redband trout are not as closely related to Sheepheaven

redband as are those in Trout Creek (Nielsen 1999).

METHODS

Abundance Sampling

Trout and Tate creeks were surveyed using a modified version of the Hankin and Reeves (1988) protocol during July/ August of 2008 and June/ July of 2009. Fish were sampled and handled under Humboldt State University Institutional Animal Care protocols (Use Permit Number 07/08.F.02.A). The Hankin and Reeves methodology involves censusing fish by single pass dive counts from a large primary sample of each habitat strata. Secondary phase units are randomly chosen from primary selected units and fish within them are sampled using multiple-pass electrofishing surveys. The ratio of fish counts from the average of secondary, electrofished units to fish counts in primary, single pass dive units is used to calibrate primary units to the more intensive survey method. Fish estimates are then extended to all units of a given habitat strata in the survey reach. D. Hankin (personal communication, 2009, Humboldt State University, 1 Harpst Street, Arcata, CA 95521) recommended two modifications to the Hankin and Reeves protocol. The first was random selection of units that were stratified by distance (i.e.units were randomly selected from a given subset rather than from the pool of total units) thus allowing for a more even distribution of units throughout each study reach. The second was substitution of bounded counts dive estimation for electrofishing calibration in pools containing 20 individuals or less. As none of the calibrated pools on either stream were observed to contain more than 20 fish on the initial dive pass, bounded counts dive estimation was routinely followed. Two habitat strata were distinguished by

current velocity: fast (erosional bed forms at low flow with broken water surface), and slow (depositional bed forms at low flow with tranquil water surface). Riffles were surveyed exclusively by electrofishing.

This modified version of the Hankin and Reeves methodology (1988) is desirable for use on small streams with threatened or endangered species in that it relies less extensively on electrofishing, thereby reducing injury and mortality. It also produces estimations of greater accuracy, compared to the standard Moran-Zippen removal method estimators, when working with small populations (personal communication, D. Hankin, 2008. Department of Fisheries Biology, Humboldt State University, 1 Harpst Street, Arcata, CA 95521). Because of CDFG concerns over possible fish mortality induced by electrofishing, the modified Hankin and Reeves protocol was altered further by calibrating electrofish units. Instead of subjecting all "fast" habitat strata to multiple pass electrofishing, primary selected units were electroshocked using a single pass and secondary phase units received two passes.

Fish surveys took place in three phases: (1) Stream reaches were habitat-typed and units randomly selected for future sampling; (2) Divers returned to snorkel pool habitat units randomly selected at the primary and secondary phase; (3) Selected riffle habitats were electroshocked according to a calibration method similar to that for pools with a primary and secondary phase of sampling. The majority of work was completed with a two-person crew. In July of 2009, the electroshocking crew on Tate Creek was augmented with one volunteer.

Habitat Mapping

Habitat mapping began on Trout Creek at the Trout Creek USFS campground (latitude: 41° 26' 57.229" N, longitude: 121° 53' 29.160" W) and proceeded upstream for 4.45 km, until individual habitat units were too small to conduct accurate snorkel observations. Mapping on Tate Creek began upstream of a large beaver pond (latitude: 41° 14' 44.616" N, longitude: 121° 55' 33.147" W) and proceeded upstream for 8.2 km.

Habitat types were designated as riffle or pool. Dominant features of riffle (fast) habitat included steep gradient, high velocity, coarse substrate, chaotic, standing waves, and frequent exposure of substrate material in the thalweg. Pool (slow), units were defined as having shallow gradient with low velocity, level hydraulic topography, fine substrate, and large eddies. Runs, lacking the hydraulic scour characteristic of pools, but similar in other attributes, were also categorized as "slow" type units. Stream sections that could not be accurately surveyed (cascades, shallow side channels, beaver ponds, flow trickling through dense brush) were removed from the sampling universe. Unit breaks were located at transitions in channel form. Units not exceeding their average wetted width in length were lumped with the most similar adjacent upstream or downstream unit. Surveyors recorded two representative widths, total length, and maximum depth at each unit. Temperature was measured with a hand-held thermometer periodically throughout each survey day. Large woody debris was tallied for pieces within bankfull width greater than three meters in length and larger than 15 centimeters in diameter at the large end (Harig and Fausch 2002). Rootwads were counted as one piece of large woody debris.

Units of each type were selected for fish census using a random stratified method. Primary units were selected at a rate of 10 % and secondary units at 25 %, per CDFG permit requirements, within randomly selected quarter mile reaches in 2008. This process resulted in too few calibration units and abundance estimate failure. In 2009, the number of primary selection units needed to produce the minimum calibration units for a statistically valid abundance estimate was determined for each stream based on 2008 habitat surveys. Trout Creek was then surveyed with the same primary selection rate (one unit randomly selected from each set of 10) but applied to the entire section of stream in 2009. Tate Creek's primary sampling rate was set at seven % (one unit randomly selected from each set of 14) and also applied to the entire section of stream surveyed in 2009. Secondary phase unit selection was set at 25% (one randomly selected out of every four primary units) on both streams.

Snorkeling Protocol

Crew members returned to primary phase selection pools located with the aid of a GPS device and flags placed at each unit during habitat mapping surveys. Prior to snorkeling, divers calibrated length estimates by viewing predetermined lengths of wooden dowel, underwater, in an unselected unit. To minimize fish disturbance, one crew member stood well back from the pool and took data communicated orally from the diver. Each diver began fish observations at the downstream end of a given unit and proceeded upstream. Fish were identified by species and total length was estimated to the nearest ten millimeters. In 2008 fish tallied by direct observation snorkeling were placed in 50 millimeter size categories. If a pool unit was also selected for secondary phase sampling and no more than 20 redband trout were observed, three additional dive

passes were made. A bounded counts dive estimator was used to estimate the number of fish present as:

$$\tilde{y}_B = d_{[m]} + \left(d_{[m]} - d_{[m-1]}\right)$$

where $d_{[m]}$ is the highest dive count and $d_{[m-1]}$ is the second highest dive count in unit *B*. Bias adjustment was unnecessary in dive units because observation probability was always well above 0.7 (personal communication, D. Hankin 2008. Department of Fisheries Biology, Humboldt State University, 1 Harpst Street, Arcata, CA 95521). Dive counts were repeated on both streams in September of 2009.

<u>Spotlighting Protocols</u>

Reconnaissance efforts on both streams revealed redband trout in extreme upper reaches that would be excluded from sampling under the modified Hankin and Reeves approach and which could not be electroshocked because of permit restrictions. Trout abundance in these areas, as well as in a main tributary to upper Trout Creek, was assessed by spotlight survey following the methods of Hickey and Closs (2006). Surveys began at sunset and progressed upstream until dense brush prevented observation. The wetted channel was scanned by two crew members with spotlights from the thalweg to their respective sides of the bank. To minimize disturbance surveyors remained outside of the channel whenever possible. Observed fish were identified to species and placed in 50 millimeter size categories. Small channel size and lethargic nocturnal behavior of fish facilitated observation.

All pool and riffle units sampled in the reach surveyed by the modified Hankin and Reeves approach were also spotlighted to compare abundance estimates generated under spotlighting and Hankin and Reeves methods. Spotlighting was always conducted prior to electroshocking as it is a relatively noninvasive method and less likely to induce fish movement (Hickey and Closs 2006). Possible linear relationships between Trout Creek abundance data from snorkel and electroshocking units (straight counts observed in primary selected units and estimated abundance in secondary phase units), and spotlighting methods were evaluated with a Pearson correlation test. Pools and riffles were assessed separately.

A consistent positive relationship between dive and spotlighting counts in pools was observed (correlation coefficient = 0.62, P = .02, n = 14). In riffles, however, electrofishing abundance estimates and spotlighting counts were inconsistent (correlation coefficient = 0.22, P = 0.46, n =14). Because conversion between spotlighting and both snorkel and electroshock estimates could not be achieved, spotlighting survey results in pools and riffles of the upper reaches on both streams were reported as direct counts of numbers observed.

<u>Electrofishing Protocol</u>

I selected riffles for electrofishing in the same manner and at the same rate as pool units. Block nets were first placed at the bottom of each selected riffle unit to prevent downstream movement of disturbed fishes. Primary units were exposed to a single pass and secondary phase units received two passes per CDFG permit requirements.

Fish captured during the first pass of a calibrated riffle were placed in a covered bucket with a battery-powered aerator and assessed following the second pass. Abundance ($\hat{y}_j^*$) in calibration units was estimated as:

$$\hat{y}_j^* = \sum_{i=1}^{r-1} c_i + \frac{c_r}{\hat{q}},$$

where c_i is the number of fish captured on pass i, c_r is the number of fish captured on the final pass, and capture probability ($\hat{q}$) is determined by:

$$\hat{q} = \frac{\bar{c}_1 - \bar{c}_r}{\sum_{i=1}^{r} \bar{c}_i - \bar{c}_r},$$

where $\bar{c}_1$ is the average catch on pass one for all units and $\bar{c}_i$ is the average catch on pass i, for all units.

<u>Area Adjusted Total Abundance Estimation</u>

The modified Hankin and Reeves abundance estimate method was altered to accommodate CDFG permit limitations on riffle unit calibration. The modified Hankin and Reeves methodology allows for incorporation of electrofishing in pool abundance estimation by substituting a bias-adjusted jackknife estimator and associated mean standard error for the pool (bounded dive counts) estimator if more than 20 fish are observed. The same area adjusted abundance estimator for pools was utilized for riffle

units exclusively with jackknife estimators and associated mean standard error (MSE)
substituted accordingly. Single pass electroshock units were treated as single pass dive
units. Abundance in pools was calculated according to the standard method. The
following notation is utilized:

$$\hat{Y}_{DA} = \text{total abundance in a given stratum}$$

$$N = \text{total number of units in stratum}$$

$$\bar{\bar{y}}_2 = \text{mean estimated abundance of phase 2 units}$$

$$\bar{x}_1, \bar{x}_2 = \text{mean first pass} (\tilde{y}_B^* \text{ or } \tilde{y}_J^*) \text{ count for phase 1 and phase 2 units}$$

$$\overline{A} = \text{mean area of units in a given stratum}$$

$$\bar{a}_1, \bar{a}_2 = \text{mean area of phase 1 and phase 2 units}$$

$$f_1, f_2 = \text{sampling fraction for phase 1 and phase 2 samples}$$

$$n_1, n_2 = \text{sample size for phase 1 and phase 2 samples}$$

$$\tilde{y}_{2k} = \text{estimated abundance} (\tilde{y}_B^* \text{ or } \tilde{y}_J^*), \text{in phase 2 sample unit k}$$

$$a_{1k}, a_{2k} = \text{area of phase 1 or phase 2 unit } k$$

$$\bar{\bar{y}}_2 = \text{mean estimated abundance of phase 2 units}$$

Total abundance in each stratum was determined with:

$$\hat{Y}_{DA} = N\bar{\bar{y}}_2 \left(\frac{\bar{x}_1}{\bar{x}_2} + \frac{\overline{A} - \bar{a}_1}{\bar{a}_2} \right)$$

The sampling variance of estimator $\hat{Y}_{DA}$ was calculated with:

$$\hat{V}*(\hat{Y}_{DA}) = N^2(1-f_1)\left(\frac{\overline{A}}{\overline{a}_1}\right)^2 \frac{s_{\tilde{y}|a}^2}{n_1} + N^2(1-f_2)\left(\frac{\overline{x}_1}{\overline{x}_2}\right)^2 \frac{s_{\tilde{y}|x}^{2*}}{n_2},$$

where,

$$s_{\tilde{y}|a}^2 = \frac{1}{n^2-1}\sum_{k=1}^{n_2}\left(\tilde{y}_{2k} - \overline{\overline{y}}_2\frac{a_{2k}}{\overline{a}_2}\right)^2$$

$$s_{\tilde{y}|x}^{2*} = \frac{1}{n_2-1}\sum_{k=1}^{n_2}\left[MSE(\tilde{y}_{2k}) + \left(\tilde{y}_{2k} - \overline{\overline{y}}_2\frac{x_{2k}}{\overline{x}_2}\right)^2\right]$$

and $MSE(\tilde{y}_{2k}) = \left(d_{[m]2k} - d_{[m-1]2k}\right)^2$ if $\tilde{y} = \tilde{y}_B$ (pools)

and $MSE(\tilde{y}_{2k}) = r(r-1)c_{r2k}$ if $\tilde{y} = \tilde{y}_j$ (riffles)

Fish Marking

Fish captured in riffles were anesthetized using a 40 mg/L solution of Finquel®
(tricaine methanesulfonate) in fresh water. Because so few fish were collected in 2008,
additional individuals were obtained by electrofishing in units dispersed throughout each
stream. Four such units were located in Trout Creek and two in Tate Creek. Captured
fish were identified to species, measured to the nearest millimeter (FL), and weighed to
the nearest 0.01 grams. In 2008 fish equal to or greater than 70 mm FL were marked
with 8-mm passive integrated transponder (PIT) tags. The alphanumeric code of each tag

was recorded along with the weight and length of each fish. PIT tags were injected into the abdominal cavity through a small incision on the ventral surface between the posterior tip of the pelvic fin and the interior point of the pelvic girdle with a 12- gauge syringe. PIT tags and syringe were sterilized in a 70 % ethyl-alcohol solution prior to use. Incisions were sealed with cyanoacrylate adhesive. In 2008, 28 and 25 redband trout were tagged in Trout Creek and in Tate Creek, respectively. All fish captured in 2009 were scanned for the presence of PIT tags.

<u>Scale Collection and Analysis</u>

In 2008, scale samples were taken from redband trout greater than 70 mm fork length to assess age structure and growth rates. Scales were removed with a knife from the left side of the fish between the dorsal fin and lateral line. The knife was thoroughly cleaned after each scale removal to avoid contamination of samples.

A minimum of ten scales per sample was mounted between two microscope slides. Scale circuli, which are concentric rings indicating growth, condense as growth rate slows during winter months. Inter-circuli spacing then widens during the spring as growth rate increases. For fish spawning in the spring, the terminus of circuli constriction represents annulus formation equivalent to one year of growth. Using Image Pro Plus^{TM} software, measurements between annuli were made sequentially from the scale focus outward to the margin of each annulus and to the outer edge of the scale. Each annulus was observable by the tightening of inter-circuli spacing. The relationship between individual fish length and total scale radius was analyzed with linear regression as:

$$\hat{L}_c = (\hat{a} + stream\_i) + \hat{b}S_c$$

where $\hat{L}_c$ is length at capture, $\hat{a}$ is line intercept, $\hat{b}$ is the regression coefficient of the

linear regression of scale radius on fork length, S_c is total scale radius, and i is 1, 2, where

i=1 is Tate Creek and i=2 is Trout Creek. The length-scale relationship from both

streams was analyzed with ANCOVA. The interaction term (radius*stream) was found

to be insignificant (P=0.381, df = (1,46), n = 50, F statistic = 0.78) and removed from the

regression. The resulting average coefficient was then adjusted to produce biological

intercepts ($\hat{a}$ - the length at which scale formation begins) for each stream.

The Fraser (1916) and Lee (1920) equation was used to back-calculate lengths at

previous ages:

$$\hat{L}_i = \hat{a} + (L_c - \hat{a})\frac{S_i}{S_c}$$

where S_c is total scale radius, S_i is scale radius to annulus i, and L_i is length at age

(annulus) i.

Grand mean back-calculated lengths at age were calculated using a weighted

average of mean back calculated lengths at age for individual cohorts. Confidence

intervals for each length-at-age estimate were constructed with one sample t-tests.

Relative Weight and Growth Rate

Relative weight analysis can be an indicator of fish health and prey abundance and allows for comparison of condition across length classes, species, and populations (Murphy et al. 1990). Relative weight (W_r) of an individual was determined with:

$$W_r = 100 * W / W_s$$

where W is the weight of an individual sampled fish and W_s is the length specific standard weight of the species. Standard weight (W_s) was calculated using the standard-weight equation developed by Simpkins and Hubert (1996) for lotic populations of rainbow trout in North America:

$$\log_{10} W_s = -5.023 + 3.024 \log_{10} TL$$

where W_s is the standard weight in grams and TL is the total length in mm. Total lengths (TL) of fish measured to fork length (FL) in the field were obtained using the conversion provided by Simpkins and Hubert (1996):

$$TL = -.027 + 1.072 FL$$

Estimated mean length for each age group, derived from scales, reflect accumulated growth between years. The growth rate of a specific cohort for a given year was calculated using Ricker (1975):

$$g = \frac{\ln L_{i+1} - \ln L_i}{t} \times 100$$

where L_i is back calculated length at age i, and t is 365 days. Growth rate was also reported as change in average length in millimeters between age classes.

<u>Minimum Stream Length</u>

Minimum stream length (MSL) was estimated following Hilderbrand and Kershner (2000) as mentioned earlier in text. Because abundance estimates include a measure of uncertainty, densities were also calculated at the upper and lower bounds of each confidence interval. One minus s reflects the combination of survival and immigration rate. Because results were not obtained from all habitat types and reaches for each sampling event and USFS surveys were conducted by different methods in previous years, s could not be empirically attained with certainty. The range of values in Hilderbrand and Kershner (2000) for proportion of fish exiting the population was used instead to estimate space requirements over a set of possible values of s at 10, 30 and 50%.

Previous Forest Service data on densities of redband trout in lower Trout Creek reported at individual sites (United States Department of Agriculture 2008), were

averaged with the 2009 upper Trout Creek data and used with Hilderbrand and

Kershner's (2000) range for *s* to calculate a rough minimum stream length estimate for

the combined stream length of Forest Service and 2009 surveys. Data and stream length

from Forest Service Trout Creek site number six were not included as this site was

dewatered by plug-and-pond restoration activities in 2006 (United States Department of

Agriculture 2008). Forest Service data on Tate Creek were not combined because only

two sites were surveyed in the lower reach by the Forest Service and my survey covered a

larger portion of the stream.

<div align="center">Culvert Analysis and Natural Barriers</div>

Existing culverts (Figures 2, 3) were assessed using FishXing survey protocols

(Furniss et al. 2006). Each culvert was rod-and-level surveyed to measure total gradient,

jump height, exit pool depth and length, exit pool tail depth, distance from maximum

pool depth to culvert outlet, distance to rest position above the culvert, and rust-line

height to the nearest tenth of a foot. Data were then analyzed with FishXing software.

FishXing estimates of passage success were found to be conservative by Burford

et al. (2009) and compensatory modifications to software input were made according to

recommendations by M. Lang (personal communication, 2009. Department of

Environmental Resources Engineering, Humboldt State University, 1 Harpst Street,

Arcata, CA 95521). FishXing swim speeds are based on flume studies and are much

reduced relative to swim speeds observed in the field. Burst and sustained swim speeds

were calculated according to Gallagher (1999) and entered under "User-defined Swim

Speeds" as 2.84 and 1.65 meters per second for large adult redband trout ($\geq$ 235 mm FL)

and 0.6 and 0.35 meters per second for juveniles ($\geq$ 50 mm FL). Maximum leap speeds

were set equal to burst speeds calculated from Gallagher (1999). Velocity reduction factors at the inlet, barrel, and outlet were decreased from one to 0.4 to account for substrate roughness in open-arch culverts. FishXing fish passage models were run at flows ranging from an extreme low of 0.03 cubic meters per second (m3s-1) to well beyond bankfull discharge at 2.83 m3s-1 on both streams.

Hydrograph data were not available on Trout or Tate creeks and duration of impassable flows was therefore impossible to calculate. However, rust line height within the culvert is a good indicator of water surface elevation at annual high base flow. In high elevation, snow dominated systems high base flow typically occurs during spring snow melt and provides a benchmark for discharge that may impede fish passage in culverts on a seasonal basis. High base flow was determined by running FishXing models at various rates of discharge until the rust line height matched the water depth at the culvert midpoint. (Flow constriction typically creates locally increased depth and lower velocity at the culvert entrance).

Potential natural barriers were noted during habitat mapping surveys. The U.S. Forest Service also provided information on effectiveness and duration of percolation barriers on Trout Creek (personal communication, S. Bachmann 2008. Shasta-Trinity National Forest, 2019 Forest Road, McCloud, CA 96057).
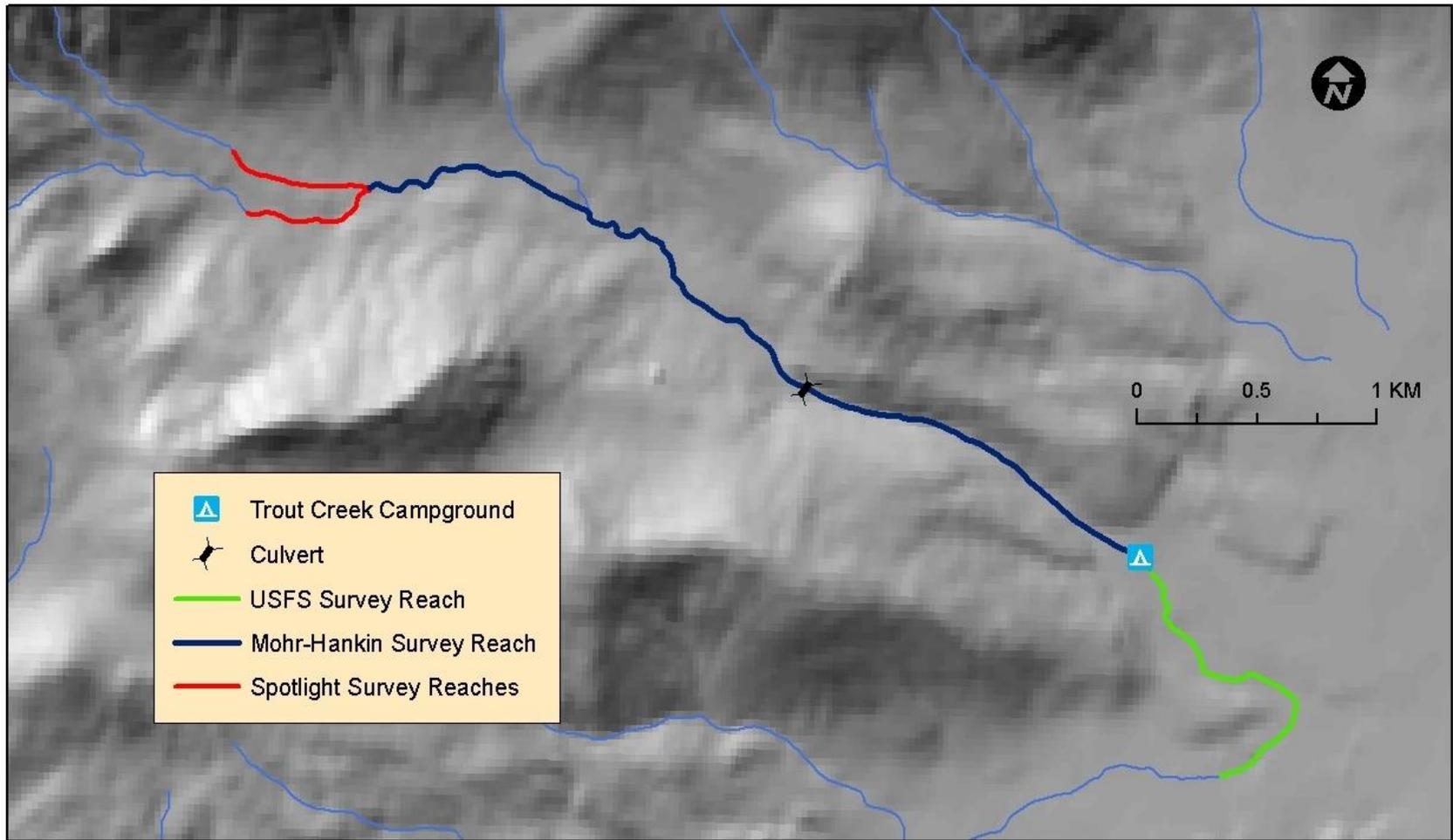
Figure 2. Culvert location and reaches in Trout Creek that were sampled by the USFS and in 2009 using Mohr-Hankin and spotlighting methods for estimating abundance of McCloud redband trout.
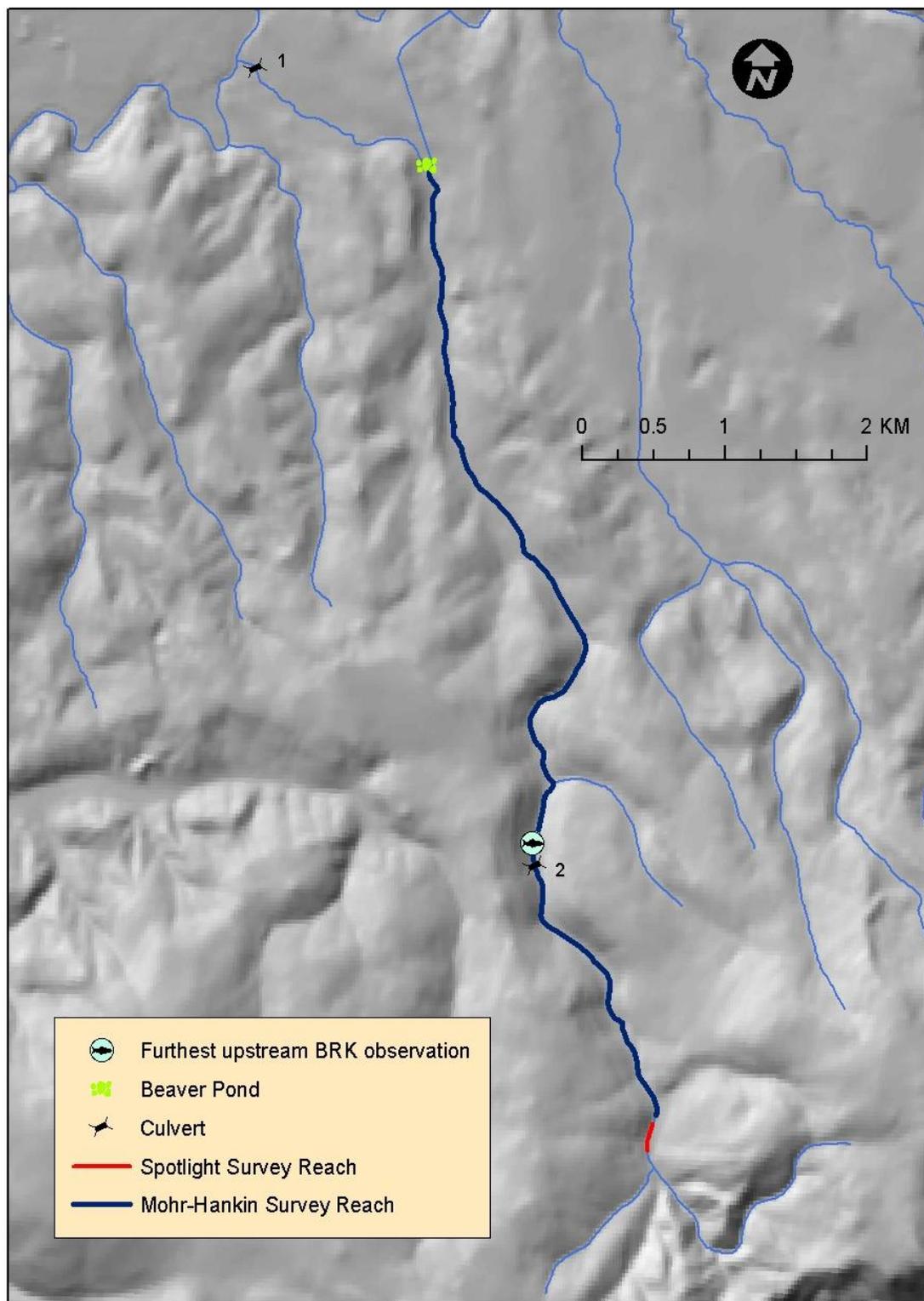
Figure 3. Culvert location and reaches in Tate Creek sampled using Mohr-Hankin and spotlighting methods for estimating abundance of McCloud redband trout, 2009.

RESULTS

Habitat Mapping

Trout and Tate creeks were of similar gradient, and exhibited similar pool/riffle ratios across both years (Table 1), but differed in other features. Habitat unit area, for example, was larger on Tate than on Trout Creek (2009 data, two-tailed t-test, df = 665, $P < 0.01$). In addition, Tate Creek discharge fluctuated less between June and September measurements, and summer temperatures were warmer than that of Trout Creek. Frequencies of deep pools and accumulations of large woody debris were greater in Trout than in Tate Creek. In both streams, average maximum depth increased by slightly more than 0.1 meters from 2008 to 2009 as 2009 surveys took place earlier in the year at slightly higher discharge (two sample t-test. Trout Creek: $P = <0.01$, df = 541, Tate Creek: $P = <0.01$, df = 836).

Abundance

Sufficient numbers of calibration units could not be attained in 2008 at a sampling rate of 10 % within truncated quarter mile reaches. Abundance estimation results from 2008 were either incomplete or statistically unrepresentative. This was because abundance in Tate Creek pools was based only on the five primary phase and two calibration units that could be selected in surveys limited to two quarter mile reaches. Stream-wide comparison of seasonal changes in abundance was not possible because permit requirements prevented repeat electrofishing of riffles within the same year.

Table 1. Habitat parameters measured on Trout and Tate Creeks in 2009 (unless otherwise indicated). Discharge was measured June 16 and September 14 on Trout Creek and June 24 and September 15 on Tate Creek Stream temperatures were observed June 9-10 on Trout Creek and June 25-29 on Tate Creek.

Stream	Pool/ Riffle ratio		Habitat mapped (km)	Deep pools/km (≥ 0.76 m)	Q (m^3s^{-1})	Temperature (C)	Gradient (%)	LWD/km	$\bar{x}$ Unit area (m^2)
	2008	2009							
Trout	1.12	1.10	4.45	9.6	.24 -.07	5 – 8.5	3.8	72	67.1
Tate	1.07	1.12	8.2	4.9	.29 -.24	8 – 12.5	4.2	43	88.4

In addition, spotlighting of upper stream reaches was not conducted until 2009. Results are therefore focused on the 2009 June/ July sampling efforts for both streams.

The redband trout population estimate for the upper 4.45 km of Trout Creek surveyed in June of 2009 was 1,969 ± 78 with a density of approximately 442 redband trout per km of stream. The Tate Creek redband trout population in the 8.2 km reach of stream sampled in July of 2009 was considerably larger with an estimate of 5,557 (± 95) and density near 678 redband trout per km. Abundance estimates by method, date, and habitat type for each stream are given in Table 2. Slightly more redband trout were estimated to occupy riffles than pools in Trout Creek while the reverse was true in Tate Creek. Pool abundance estimates increased in Trout Creek from June to September sampling efforts while the reverse was found in Tate Creek.

Approximately 0.84 kilometers of upper Trout and 0.37 kilometers of upper Tate Creek were spotlighted. Counts roughly doubled in both streams from June to September (Table 2). Total numbers of redband trout observed by spotlighting in upper reaches were much less than abundances in lower reaches estimated using modified Hankin and Reeves methods, but some of the largest individuals (> 300 mm FL) were observed in the upper reaches of both streams. Fish per kilometer decreased to 378 km^{-1} and 652 km^{-1} on Trout and Tate creeks respectively, when spotlight surveys were included in density calculations. Observations of redband trout in spotlighted riffles selected by the modified Hankin and Reeves method in Trout Creek increased from five in June to 14 in September. Trout Creek's northern tributary may be devoid of redband trout. None were seen in 0.70 kilometers of stream surveyed in June and September and the stream was nearly dry in September.

Table 2. Redband trout abundance estimates by method, date, and stream. The Mohr-Hankin estimator failed due to lack of calibration units in Trout Creek in July 2008, and in riffles in Tate Creek in August 2008. Abundance estimates are not available from riffles in September 2009 for either stream because only one electrofish effort was allowed annually by CDFG. Upper reaches were not surveyed in 2008.

	Mohr-Hankin					
	Trout Creek			Tate Creek		
Survey Date	July 08	June 09	Sep 09	August 08	July 09	Sep 09
Pools	NA	940 ± 13	1422 ± 45	2572 ± 10	3147 ± 58	1663 ± 59
Riffles	NA	1029 ± 65	NA	NA	2410 ± 37	NA
95% C. I. for total estimate	NA	1,891-2,047	NA	NA	5,462-5,652	NA
	Spotlight Only					
	Upper Trout Creek			Upper Tate Creek		
Pool and Riffle Total	NA	33	53	NA	24	68

Young of the year (YOY: 25-35 mm FL) were not observed until September by snorkel and spotlight surveys in both streams. Brook trout were not observed above pool 148 (latitude: 41° 12' 12.002" N, longitude: 121° 54' 57.087" W) on Tate Creek. Pool 148 is located approximately 150 meters downstream of Tate Creek culvert number two (Figure 3) and roughly marks the beginning of the high-gradient upper reach. Brook trout were not encountered in Trout Creek. Brown trout were not observed in either stream.

Fish Marking

A single PIT-tagged redband trout was recovered in Trout Creek in 2009 and no marked fish were recaptured in Tate Creek. The recaptured redband trout exhibited growth from 104 mm FL and 13.46 g to 120 mm FL and 25.04 g over one year.

Scale Analysis and Growth Rate

Scale analysis proved difficult due to the paucity of scale circuli and consequent lack of obvious annuli formation. Of 27 Trout Creek scale samples, eight were rejected for analysis because of excessive regeneration or physical damage. Twelve Trout Creek redband trout, ranging in length between 85 and 195 mm, were identified as age 1+ and seven (length range: 153 - 215 mm) as age 2+. Of 32 Tate Creek scale samples, nine were dropped from analysis because of poor quality. Twenty Tate Creek redband trout (length range: 71 – 129 mm), were aged at 1+ year; three (length range: 174 – 219 mm) were aged at 2+ years. Age 2+ was the maximum observed age for redband trout on both streams. Biological intercepts ($\hat{a}$), calculated for use in the Fraser (1916) and Lee (1920) equation, for Trout and Tate creeks were 55.77 and 28.03 respectively with an adjusted R

squared of 66% on the average coefficient of the regression of body length on scale radius. Trout Creek produced larger fish at age than Tate Creek but growth rate in Tate Creek exceeded that of Trout Creek redband trout according to the Ricker (1975) method (Table 3). However, regression slopes of the relationships between fish length and scale radius did not differ between streams (ANCOVA, P=0.381, df = (1,46), n = 50, F statistic = 0.78). Confidence intervals for back calculated lengths at age of fish from both streams were large, with length estimates for age 2 fish exhibiting greater variance than age 1 fish (Table 3). Scale samples from the only recaptured redband trout (aged 1+ in 2008) did not display formation of any annuli.

Age class structure of redband populations was not apparent from length-frequency plots (Figure 4). Fork lengths of redband trout (mm) ranged from 43 to 234 on Trout Creek and 49 to 219 on Tate Creek.

Relative Weight

Average relative weight of redband trout in Trout Creek was 85.9% and 95.9% of standard weight in 2008 and 2009, respectively. Tate Creek redband trout average relative weight was 92.9% and 92.4% of standard weight in 2008 and 2009, respectively. Difference in average relative weight between Trout Creek and Tate Creek was found to be significant (two sample t-test, P = 0.04, df = 54, n = 56, T value = 2.06) in mid-summer surveys of 2008 but not during earlier season surveys of 2009 (two sample t-test, P = 0.15, df = 149, n = 151).

Table 3. Back-calculated lengths at age and growth rate of redband trout from Trout and
Tate creeks. Growth rate was estimated following Ricker (1975).

| Stream	Age class	$\hat{L}_i$ [a] at age	95% CI	n	g	$\overline{\Delta}$ in length (mm)
Trout	1	109.4	102.1 – 116.7	19		
					0.10	47.7
	2	157.1	132.1 – 182.1	7		
Tate	1	75.5	65.7 – 85.3	23		
					0.17	62.9
	2	138.4	120.4 -156.3	3		
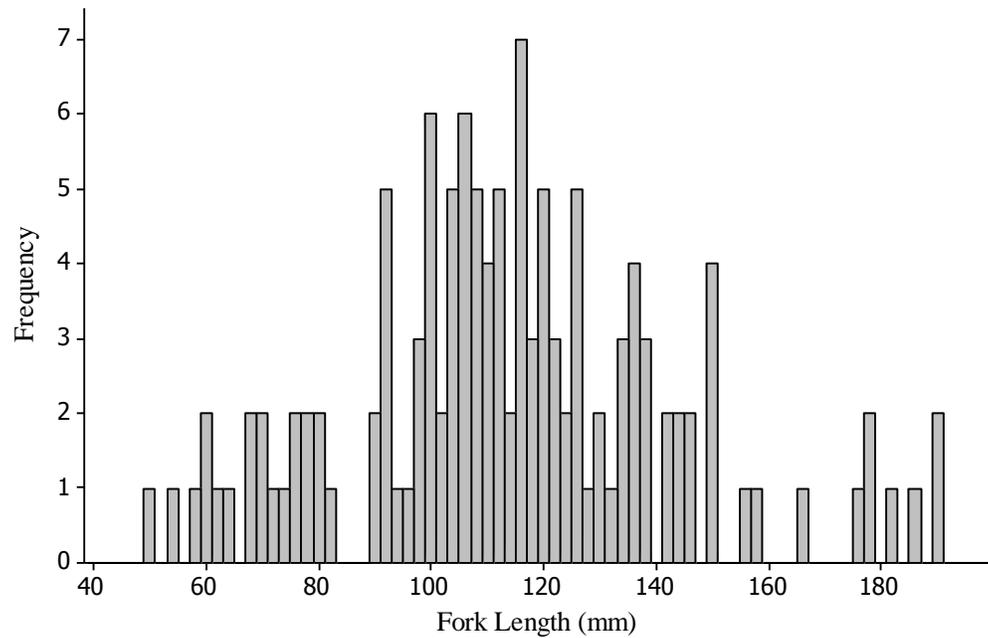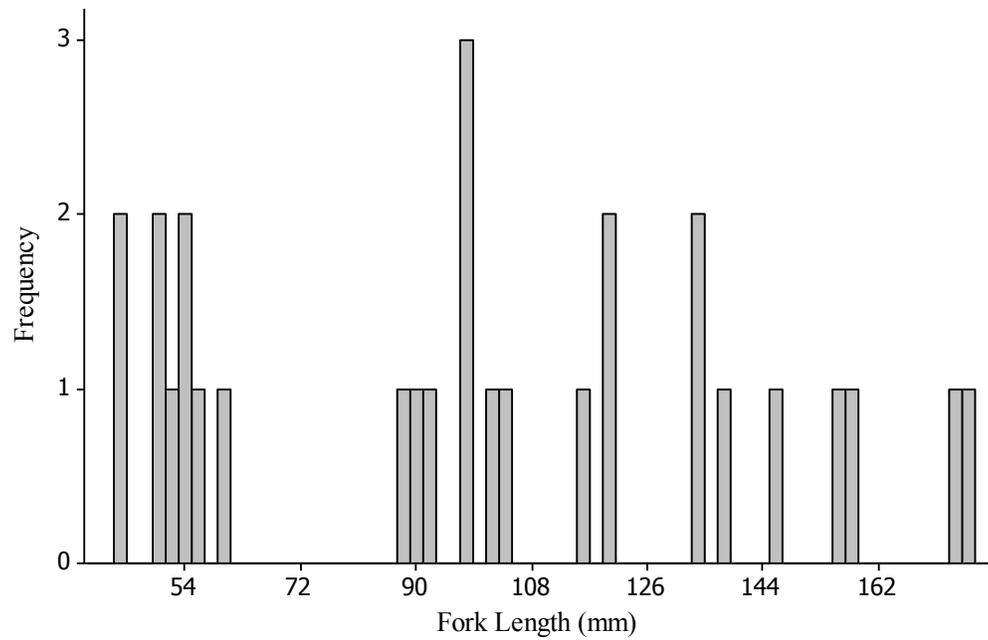
Figure 4. Length frequency histograms of redband trout captured in Trout Creek (top) and Tate Creek (bottom) in Siskiyou County, California, 2009.

Culverts and Natural Barriers

The sole Trout Creek and lower Tate Creek (number one, Figure 3) culverts are of open-arch design. Passage through these culverts is compromised only by flow constriction during very high discharge events. Neither was estimated to create a barrier for adult redband trout even at 2.8 m^3s^{-1}, which is well above high base flows of 0.71 and 1.56 m^3s^{-1} calculated from culvert rust line heights on Trout and Tate creeks respectively. The Trout Creek culvert and Tate Creek culvert number one were estimated to represent velocity barriers to juvenile redband trout at flows 0.21 m^3s^{-1} less than high base discharge. The channel through both culverts does not differ from local gradient or substrate. Variation in available velocities is high in natural channels and these culverts may be passable to juveniles at higher flow (personal communication, M. Lang 2011. Department of Environmental Resources Engineering, Humboldt State University, 1 Harpst Street, Arcata, CA 95521). The upper pipe-arch culvert (number two, Figure 3) on Tate Creek was installed at a slope slightly less than stream grade, creating a vertical drop of 0.19 meters at the outlet. This created a leap barrier for adult redband trout at flows greater than 0.31 m^3s^{-1}, well below calculated high base flow of 0.91 m^3s^{-1} at the culvert's position high in the watershed. Juvenile redband trout passage was extremely limited here (Table 4).

Structural complexity prevented empirical determination of passability at natural barriers on both streams. Vertical falls, impassable at low flow, may not constitute barriers at higher flows as side channels are inundated and jump heights reduced with increased discharge. The only plausible method of natural barrier evaluation, involving

Table 4. Range of passable flows at each culvert based on FishXing analysis based on flows ranging from 0.03 to 2.83 m^3s^{-1}. Tate Culvert #2 represents a leap barrier beyond the upper end of given range. Tate Culverts #1 and #2 represent velocity barriers beyond the upper end of given range. All discharge units are given in m^3s^{-1}.

	Culvert		
	Trout	Tate #1	Tate #2
Adult passable flow range	0.03-2.83	0.03-2.83	0.03-0.31
Juvenile passable flow range	0.03-0.5	0.03-0.54	0-0.01
High base discharge	0.71	1.56	0.91

mark and recapture of fish above and below barriers to observe movement, was prevented by permit restrictions. A percolation barrier exists downstream of the USFS abundance survey reach on Trout Creek (Figure 2) (personal communication, S. Bachmann 2008. Shasta-Trinity National Forest, 2019 Forest Road, McCloud, CA 96057). This barrier consists of several dry kilometers of streambed and can only be overcome during extreme high flow. Its exact position varies with discharge. Other possible natural barriers included numerous small cascades on both streams and a beaver dam on lower Tate Creek (Figure 3).

Minimum Stream Length

Minimum stream length was estimated at 7.35 ($\pm$ 0.29), 9.45 ($\pm$ 0.37), and 13.23 ($\pm$ 0.51) km on upper Trout Creek and 4.26 ($\pm$ 0.07), 5.48 ($\pm$ 0.09), and 7.67 ($\pm$ 0.13) km on Tate Creek using 2009 densities and s from Hilderbrand and Kershner (2000) set at 10, 30, and 50%. With the average of 2009, USFS densities, and the same values of s from Hilderbrand and Kershner (2000) minimum stream lengths were estimated at 3.65 ($\pm$ 0.03), 4.69 ($\pm$ 0.04), and 6.57 ($\pm$ 0.06) km on the combined Trout Creek reach.

DISCUSSION

Differences in abundance between McCloud redband trout from Trout and Tate creeks were likely not attributable to differences in physical habitat features. Perhaps the greater density of deep pools and LWD found in Trout than in Tate Creek did not produce increased fish abundance because they occurred in a higher elevation, colder, less productive environment. Streams lower in the watershed likely benefit from an increased availability of invertebrate prey because of upstream processing of leaf litter and higher primary production associated with reduced canopy shade (Vannote et al. 1980). Tate Creek's higher overall gradient was skewed by the upper reach (6.4 % above culvert number two). Tate's lower reaches were characterized by low gradient, consistent discharge, large individual habitat units, and reduced canopy cover with temperatures exceeding those in Trout Creek. These factors, along with Tate Creek's larger drainage area, may have enhanced invertebrate communities, nutrient retention, and fish production as reflected in abundance estimates.

The section of Trout Creek below my survey reach may offer similar habitat (low gradient, warmer temperatures) to that found in lower Tate Creek, and may explain the higher densities observed in Trout Creek by previous USFS and CDFG surveys. Forest Service surveys on Trout Creek also took place largely on the 1.6 km of stream below Trout Creek campground with only the furthest upstream USFS site overlapping 2009 surveys. Averaged over the entire length of stream, densities of fish in the two streams may be comparable.

Densities in both streams observed in 2009 were far below the average density calculated from USFS surveys from 1975 to 2005 (983 redband trout/ km and 962

39

redband trout/ km in Trout and Tate creeks respectively). Comparisons between 2009

data and USFS studies are complicated by differing sampling methods and spatial

coverage. Forest Service data were collected by point sampling and extrapolated to the

entire stream (United States Department of Agriculture 2008) while my data were

collected by stream-wide sampling that I conducted in 2009. Nonetheless, differences in

densities between my study and previous sampling efforts suggest that annual variability

in population abundance in the streams is likely to be high. Forest Service data

substantiate this. For example, between 1981 and 2005, estimated densities of redband

trout ranged between 98 to 6071 trout/ km in Trout Creek, and between 244 to 3019

redband trout/ km in Tate Creek from 1975 to 2005 (United States Department of

Agriculture 2008).

Wide fluctuations in population density undoubtedly reflect environmental

stochasticity (Ricklefs and Miller 2000) particularly with respect to streamflow. Because

of the porous, volcanic soils in the region, many streams in the upper McCloud watershed

have long dry or intermittent reaches even in years of average or above average flows.

During a period of drought from 1987-1992, flowing stream habitat in upper McCloud

shrank from 96 to 37 km (United States Department of Agriculture 2008). Seasonal

habitat shrinkage was evident in my 2009 survey of Trout Creek, and could have

accounted for the increased abundance of trout in September compared to June as fish

concentrated in smaller available area of both habitat types (Table 2). Greater fall

abundance in Trout Creek probably resulted from ease of visual observation at lower

discharge by both snorkel and spotlight methods and could also account for increases in

spotlight counts of upper reaches in the fall.

Upper reaches in both streams do not appear to represent critical habitat as very few fish were observed in these areas. Spotlight observations may have also been biased towards larger fish because smaller fish are less conspicuous (Hickey and Closs 2006). Correlation failure between spotlighted and electrofished riffles may reflect the difficulty of spotlighting observations in high velocity, turbulent habitat. The stronger correlation between spotlighting in pools and dive counts was probably caused by the difficulty of dive observations in small Trout Creek pools where fish were easily disturbed during daylight hours. Juvenile fish were often observed in the shallow tail-out portion of pools at night, habitat they would be unlikely to occupy without cover during the day. This likely drew spotlight counts closer to dive observations in some units. Hickey and Closs (2006) also reported greater spotlighting observer efficiency in slow velocity units.

Typically low discharge seems to affect the viability of redband populations in upper McCloud tributaries. However, harm from nonnative salmonids was not evident in the upper reaches of Trout or Tate creeks. The absence of brook trout and brown trout in Trout and upper Tate creeks may be explained by their differing reproductive strategies relative to redband trout. Streams in the upper McCloud basin are subject to a snowmelt-pulse dominated hydrograph. Brook and brown trout spawn in the fall and fry emergence consequently occurs during peak discharge. Weak-swimming fry are energetically taxed by a lack of velocity refuge during these flows and recruitment is limited. Brook trout have greater success in high elevation streams than brown trout because they seek out low velocity areas with warm ground water seepage for spawning (Wood and Budy 2009). FishXing analysis estimated Tate Creek culvert number two as a partial barrier yet redband trout were observed in the upstream reach, suggesting occupation prior to

culvert installation and (or) successful culvert passage. Existing environmental conditions may then be more responsible for inhibiting brook trout colonization of upper, higher gradient, Tate Creek reaches than the position of culvert number two.

Meaningful assessment of age and growth based on scale samples from both streams is probably impossible due to errors that almost certainly occurred in back calculation of lengths at age. Accurate back calculation of length at age depends on identification of periodic features on calcified structures that may vary greatly depending on individual fish or environmental conditions. Late spawning and a short growth period at high elevation may inhibit development of first year annuli. Coleman and Fausch (2007) observed cutthroat trout fry at 26-30 mm TL in September in high elevation streams. They postulated that low temperature caused delayed spawning, prolonged egg incubation, and slow larval development. Tightly spaced outer annuli in older fish may also be obscured by scale erosion (Kruse et al. 1997). These factors may be in effect on tributaries to the Upper McCloud River as young of the year (~25-35 mm TL) were not observed until early September and scales on older fish were largely unreadable some distance from the scale margin. Annuli formation is predicated on growth response to significant changes in temperature which may not occur in systems in extreme northern latitudes, high elevation, or the tropics. The paucity of total circuli produced by fish in slow-growing populations presumably reduces the visible width of annuli formation. These biological responses to extreme environmental conditions likely combine to obscure annuli and reduce age estimates.

Studies in high elevation systems have had similar difficulties in determining age from scale samples. Hubert and Baxter (1987) and Kozel and Hubert (1987) found

analysis of scales to yield reduced age estimates when compared to otoliths. Kruse et al. (1997) reported significantly less variance in age estimates based on otoliths than those based on scales. Lack of any apparent annuli on the recaptured fish suggests that age of other fish could have been underestimated. Large confidence intervals in back calculated length at age of redband trout may be explained by under estimation of age and subsequent inclusion of multiple ages in each assumed cohort class. Although lengths at age were similar to those reported by the USFS (United States Department of Agriculture 2008) for redband trout and Moyle (2002) for rainbow trout in high elevation streams, only two age classes were identified in 2009. The lone recaptured redband trout exhibited only 16 mm of growth in one year and was at least two years old by the time it reached a length of 120 mm, which is less than half the length of the largest individuals observed. The USFS (United States Department of Agriculture 2008) also reported four age classes for redband trout and Behnke (2002) described a maximum age of six to seven for redband trout. It is unlikely then that redband trout in tributaries to the Upper McCloud grow at a rapid rate and perish after two years. This hypothesis cannot be tested with scale analysis as the sole method of age and growth determination. Employment of multiple methods such as mark-recapture and otolith analysis may be needed to accurately estimate age and growth in high elevation salmonid populations.

Relative weight on both streams was consistently greater than 80% of standard weight, well within the range found in healthy populations (Anderson and Neumann 1996). Small sample size on Trout Creek (<30), lack of invertebrate sampling, and infrequent surveys prevent meaningful evaluation of possible differences in seasonal prey availability and its associated effect on relative weight (Van DeValk et al. 2008 and Liao

et al. 1995). Future studies should assess relative weight multiple times per year and explore invertebrate community composition to adequately assess possible seasonal constraints on redband trout growth.

Estimated minimum stream lengths on upper Trout Creek all exceed the portion of stream surveyed (5.29 km) and the population estimate of 1,969 individuals is less than the minimum ballpark figure of 2,500 needed to maintain an effective population size of 500. Trout Creek minimum stream length estimates for the combined reach should be viewed with caution as they are based on averages from past USFS point surveys. According to USFS surveys most of Trout Creek redband are concentrated in the lower reach. Higher densities here reduced minimum stream length to less than 5.63 km (which is the length of stream available in dry years, United States Department of Agriculture 2008), except with s set at 50%. Late season pool numbers may have increased in Trout Creek if fish in the de-watered lower reach moved into remaining upstream habitat. A barrier isolating upper Trout Creek would simply fragment already limited habitat and prevent potentially vital movement of redband trout from the lower 1.6 km during late summer. Redband trout movement in Trout Creek has not been investigated.

It is unlikely that hatchery rainbow trout from the mainstem McCloud would breach the existing Trout Creek percolation barrier by traveling through several kilometers of flooded forest during extreme high flow events. Brown trout exist in the reach above the barrier (personal communication, S. Bachmann 2009. U.S. Forest Service Hydrologist, Shasta-Trinity National Forest, 2019 Forest Road, McCloud, CA 96057) but successful colonization of upper Trout Creek is improbable given abiotic constraints relative to their life history strategy mentioned earlier. Low densities, apparent slow

growth, limited habitat, recreational fishing pressure, and the absence of a risk from non-native species render the upper portion of Trout Creek surveyed in 2009 a poor candidate for deliberate isolation. Given the intermediate threat of invasion and existence of a natural barrier effectively isolating a sparsely populated headwater reach, current Forest Service practices of physical removal of brown trout from the lower reach (personal communication, J. Zustak 2011. U.S. Forest Service Fisheries Biologist, Shasta-Trinity National Forest, 2019 Forest Road, McCloud, CA 96057) represents a practical management option versus habitat fragmentation through installation of another barrier (Fausch et al. 2009).

Tate Creek may represent a viable candidate for deliberate isolation. Surveyed habitat (8.56 km) exceeded the maximum estimate of minimum stream length of 7.67 km. Minimum stream length here is slightly less than the 8 km recommended by Hilderbrand and Kershner (2000) and 9.3 km postulated by Williams et al. (2009) but is based on higher densities observed in Tate Creek. Minimum habitat estimates on Tate Creek are probably more robust than those on Trout Creek because Tate Creek minimum stream length estimates are based solely on 2009 stream-wide surveys. Approximately 1.8 km of additional habitat is available downstream of the study reach. This area was not surveyed due to the presence of beaver ponds unsuited to the modified Hankin and Reeves method. Including the area not surveyed, total length of Tate Creek is about 10.4 kilometers, which is slightly more than the 10 kilometers necessary to maintain an effective population size of 500 as reported in Fausch et al. (2009). Lower Tate Creek exhibits habitat characteristic of a mid-order stream (Vannote et al. 1980) and should

contain sufficient habitat for successful isolation even by conservative estimates, but isolation would prevent potentially important access to the mainstem McCloud.

Tradeoffs between invasion threats and genetic risks associated with isolation on Tate Creek could be mitigated by a number of factors. Tate Creek redband trout appear to be hybridized (Figure 5) but protecting some less than "pure" populations may lower the risk of extinction for the entire purported species (Simmons et al. 2010). Samples from Simmons et al. (2010) genetics study were taken only from the lower reaches of Tate Creek. Redband trout observed higher in the watershed displayed physical traits similar to classic redband trout (personal observation). Tate Creek may therefore still contain relatively pure redband trout. Historic migratory life history strategies and genetic exchange could be re-established by translocating redband trout to the large, productive Tate Creek watershed after it has been isolated. Redband trout in Edson and Sheepheaven creeks may be feasible translocation candidates as they show little signs of introgression with hatchery trout but have low allelic richness indicative of significant inbreeding (Simmons 2010) imposed by their spatially limited habitat. Redband trout translocated to Tate Creek could intermingle genetically in a population large enough to maintain long term persistence.

Immediate protection of redband trout in Tate Creek could lead to eventual natural reconnection with other isolated populations. Reconnection of protected populations allows for reestablishment of migratory behavior (Fausch et al. 2009, Peterson et al. 2008) which is associated with increased growth, fecundity, and resistance
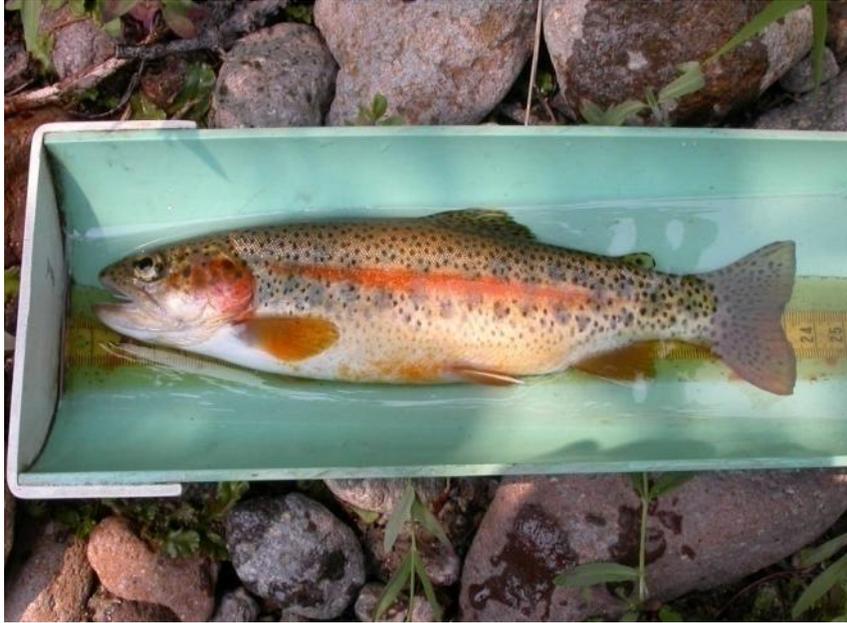
Figure 5. Redband trout captured in upper Trout Creek (top) displaying classic redband features and redband trout captured in lower Tate Creek (bottom) with morphological features similar to those of hatchery rainbow trout.

to disturbance (Peterson et al. 2008). In the meantime, isolated habitat should be augmented to maintain target populations.

Habitat improvement opportunities on Tate Creek include physical or chemical removal of brook trout and hatchery rainbow trout from reaches below the upper Tate culvert. Baffles installed in the upper culvert, or construction of an open-arch type culvert, would increase fish passage and ensure connectivity between reaches. In contrast to Trout Creek, fishing access and potential illegal stocking in Tate Creek is limited by steep, rugged terrain in the upper reach and private timber property in much of the lower reach. However, ready access to the Tate Creek – McCloud River confluence, where a barrier would presumably be located, could lead to illegal stocking of McCloud River fish in Tate Creek. Other difficulties with isolation of Tate Creek include the expense of barrier installation. Culvert number one on Tate Creek is an open-arch design and a deliberate barrier could not exploit an existing geologic knick point.

Strategy for redband trout management should include cessation of hatchery rainbow and brook trout stocking in the upper McCloud system. Fausch et al. (2009) identified propagule pressure as the main predictor of invasive species establishment. Additional mortality produced by take from sport fishing only adds to stress experienced by populations in low productivity, high elevation systems. Sport fishing should also be eliminated on upper McCloud tributaries.

Following the conceptual framework provided by Fausch et al. (2009), Tate Creek redband trout merit isolation for four reasons: (1) Tate Creek redband trout have conservation value even if populations in the lower reach have been somewhat compromised by hybridization; (2) The presence of brook trout and evidence of hatchery

rainbow trout hybridization in the lower reach indicate a high threat level; (3) Sufficient

habitat exists to maintain genetic integrity and resistance to environmental and

demographic stochastic events; (4) Tate Creek should receive high management priority

because of reduced exposure to hillslope failure from Mount Shasta and extensive

contained habitat suitable for growth of translocated populations. Future Tate Creek

studies should address annual stream-wide population fluctuation, fish movement relative

to barriers, genetics of redband trout in upper reaches, monitoring of flow and

temperature regimes, and identification of an effective deliberate barrier location before

management proceeds.

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