

SUMMER ABUNDANCE AND DISTRIBUTION OF JUVENILE
CHINOOK SALMON (*Oncorhynchus tshawytscha*) AND STEELHEAD TROUT
(*Oncorhynchus mykiss*) IN THE MIDDLE FORK SMITH RIVER, CALIFORNIA

by

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A Thesis

Presented to

The Faculty of Humboldt State University

In Partial Fulfillment

of the Requirements for the Degree

Master of Science

In Natural Resource Management: Fisheries

October 1, 1995

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95/FI-322/12/14

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ABSTRACT

Abundance of juvenile chinook salmon (*Oncorhynchus tshawytscha*) and yearling or older juvenile steelhead trout (*Oncorhynchus mykiss*)(steelhead >1) in the Middle Fork Smith River, California (sixth order stream with average summer flow of 8.5 m³/s) was estimated on two occasions during the summer of 1993 and on three occasions during the summer of 1994 using visual estimates in sample habitat units. Habitat was classified with regard to subunits of pool, riffle, and run channel units (pool head, pool body, pool tail, in-pool run, riffle margin, riffle midchannel, run margin, and run midchannel) that differed in velocity, turbulence, and depth. Aerial photographs were used to measure habitat areas. For pool bodies and riffle margins the use of habitat area in ratio estimation increased the precision of total abundance estimates by 21%-84%. Mean observable chinook density ranged from 0.0002 fish/m² in riffle midchannels to 0.2085 fish/m² in pool heads. Mean observable steelhead >1 density ranged from 0.0197 fish/m² in pool bodies to 0.4716 fish/m² in pool tails. For some habitat types, chinook abundance decreased within each summer, and steelhead >1 abundance increased between years. Analysis of the variation in repeat counts within a day indicated that measurement error was high in certain situations, but not generally higher than for visual estimates of fish abundance in smaller streams. High rates of emigration by chinook relative to steelhead >1 caused higher temporal variation in the abundance of chinook per unit. Recommendations are made for monitoring annual variation in abundance of juvenile chinook and steelhead >1 in the Middle Fork Smith River.

ACKNOWLEDGEMENTS

This thesis project was supported by the U.S. Forest Service and the Institute for River Ecosystems. I would like to extend my sincere gratitude to Dr. Gordon Reeves for the initial opportunity and a thorough manuscript review, to Mike McCain for unyielding support, and to Dr. Terry Roelofs for guidance and enthusiasm. Drs. Dave Hankin, Bill Trush and Bret Harvey provided valuable consultation and manuscript reviews. For assistance in the field, I thank Jim Simondet, Baker Holden III, Hans N. Voight, J. Scott Davis, and, in particular, Jason White whose unfaltering commitment to quality was greatly appreciated. Mark Allen provided data and manuscript review. I recognize the contribution of all my fellows of the River Institute, for we learn and accomplish together. Tesilya Hanauer helped me to keep the science in perspective. This thesis is dedicated to the memory of Jack, who took me to the river each evening and stayed at my desk with me each night.

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INTRODUCTION

Mainstem river segments (mainstems) are uniquely important to juvenile anadromous fish due to their central position within drainage basins. In large basins, mainstems are used by every smolt that enters the ocean regardless of where the fish was hatched and reared. More than simple migration corridors, mainstems may have high densities of pre-smolt fish year-round (Levings and Lauzier 1991), and mainstem habitat conditions may be responsible for distinct life history traits (Prevost et al. 1993, Dambacher 1991). In several basins, most juvenile steelhead (*Oncorhynchus mykiss*) migrate from their natal streams to mainstems a year or more before smoltification (Lieder 1986, Dambacher 1991, Harkleroad and LaMarr 1992). Downstream migrating pre-smolt chinook salmon (*Oncorhynchus tshawytscha*) feed and grow as they move through mainstems of coastal Oregon rivers from spring through summer (Nicholas and Hankin 1988). Because the size upon ocean entry for steelhead (Ward and Slaney 1988) and chinook (Neilson and Geen 1986) is a critical factor in survival, the growth opportunity or habitat quality in mainstems may be an important component in the productivity of these populations.

I define mainstems as those river segments that are comprised mostly of water too deep or too fast for safe electrofishing procedures. Habitat use by juvenile salmonids in mainstems is not well studied as a consequence of this limitation. Most commonly, seining (e.g., Levings and Lauzier 1991) or trapping of downstream migrants in various

kinds of smolt traps (e.g., Lieder et al. 1986) are used to obtain habitat use information. Seining and boat electrofishing are difficult to impossible in deep rivers of relatively high gradient. Smolt traps are used to sample condition of downstream migrants, monitor trends in emigration, and estimate smolt production. Multiple smolt traps may be used to study growth and rate of migration (e.g., USFWS 1995). Smolt traps are not useful, however, in studying some fundamental aspects of habitat use such as distribution.

Direct observation (snorkeling) is a practical method for studying habitat use by salmonids in streams (Keenleyside 1962, Heggenes et al. 1990) and has been successfully applied in mainstems (Schill and Griffith 1984, Hicks and Watson 1985, Zubik and Fraley 1988). As a quantitative method, however, snorkeling is limited because visual estimates of fish abundance can be highly inaccurate (Northcote and Wilkie 1963, Rodgers et al. 1992). Snorkeling is widely applied to stream fish inventories as part of the "basin-wide visual estimation technique" (Hankin and Reeves 1988, Dollof et al. 1993), but mainstems are typically excluded from such "basin-wide" surveys because the technique uses estimates from multiple pass electrofishing to adjust biased snorkeler counts. Despite the potential for expanding basin-wide surveys to include mainstems, the utility of snorkeling in mainstems has not been thoroughly evaluated.

The methodology of using snorkelers to quantify fish abundance in mainstems confronts four problems: (1) unknown bias, (2) potentially high variability in visual estimates within individual habitat units due to deep, swift water, (3) high emigration rates that may vary distribution and abundance too quickly for assuming a fixed object of

estimation, and (4) voluminous geomorphic channel units that may not serve as a good primary sampling units, both from the standpoints of fish distribution and snorkeler capabilities.

Visual estimates with unknown bias can be made useful by qualifying the target variable as the number of observable fish. Estimates of observable fish abundance should provide a useful index of actual fish abundance if the proportion of fish unseen (proportional bias) remains constant. Proportional bias may be consistently small when temperature is optimal and fish are not utilizing cover (Hillman et al. 1992). Visual estimates should generally not be used, however, to infer differences between habitat, species, or any other groups which likely differ in proportional bias.

Estimating fish abundance in streams is analogous to two-stage sampling (Hankin 1984). Stage one involves selecting sample habitat units (primary units) from each stratified habitat type stratum. Stage two involves determining the number of fish in each of the selected habitat units. Enumerating fish in a habitat unit can be viewed as second stage sampling in that one does not expect to count a certain number whenever that habitat is examined; visual estimates vary in response to fish behavior, changing viewing conditions, and incomplete or inaccurate counting, among other factors. When estimating observable fish abundance, measurement error is equivalent to the variability in visual estimates and can be assessed using between-diver variance (Hankin and Reeves 1988) or the variation in repeat counts.

Sampling habitat units for estimation of fish abundance assumes there to be a fixed number of fish within each habitat unit and over the entire stream segment throughout the sampling occasion. This assumption is questionable for surveys of long stream segments where many days are required for sampling, and in mainstems where the cumulative effect of seaward migrants from many upstream locations may result in exceptionally high temporal variation in distribution and abundance. Migration and its effects on abundance estimation must be studied, therefore, when evaluating the validity of any sampling survey of fish abundance in mainstems.

During a 1992 pilot period, I noted conspicuous patterns of fish distribution within geomorphic channel units (i.e., pools, riffles, runs). Within pools, relatively high densities of fish were found near the upstream end (pool head), and near the downstream end (pool tail). In riffles and runs, relatively high densities of fish could be counted when swimming upstream along either bank. Different densities of observable fish, and conspicuous differences in depth and velocity indicated at least five types of habitat subsidiary to geomorphic channel units. The utility of classifying habitat according to these observations is supported by (1) the principle of stratified sampling, whereby one seeks to maximize differences in estimated means among strata and minimize variance within strata (Stuart 1968), (2) the predominance of velocity and depth as factors in habitat selection (Everest and Chapman 1972, Steward and Bjornn 1987), and (3) the potential relationship between measurement error and the variable physical attributes of depth and velocity (Heggenes et al. 1990).

This thesis attempts to both advance methodology and generate inferences regarding habitat use in mainstems by intensive study of an eight km segment of the Middle Fork Smith River (Middle Fork), a sixth order stream with average summer flows of 8.5 m³/s. Methodology and habitat use inquiry were not separated in this thesis because they are inextricably linked. Patterns of habitat use influence methodology by indicating appropriate sampling strategies, but methodology determines the reliability and structure of data from which habitat use is inferred. Also, migration affects both the reliability of abundance estimates and patterns of habitat use. The objectives of this thesis can be separated, however, into the methodological and the biological.

The methodological objectives of this thesis were to: (1) stratify the Middle fork using habitat types defined by depth and velocity, (2) determine the utility of habitat area in fish abundance estimation, (3) assess measurement errors associated with visually enumerating fish, and (4) evaluate the utility of visual estimates of fish abundance in the Middle Fork. The stratification was evaluated as an alternative to more common stratification schemes (Hawkins et al. 1993). Measurement errors were considered according to source, and were compared among habitat, species, and time. Measurement errors were also compared to those of visual estimates in other streams. The utility of abundance estimates was evaluated with regard to the total errors of estimation and temporal variation resulting from fish migration.

The biological objectives of this thesis were to: (1) examine the distribution of fish according to habitat type, (2) estimate densities of observable fish by habitat type,

(3) test for changes in abundance of fish within summer and between years, (4) plot emigration trends through one summer, (5) measure downstream migration rates, and (6) estimate growth of fish during residence in the Middle Fork. The first three objectives were subject to the validity of visual estimates of fish abundance. The biological objectives, as a whole, sought to characterize summer habitat use by juvenile anadromous fishes in the Middle Fork Smith River.

BASIN AND SITE DESCRIPTION

The study segment was located in the Smith River basin, a 1950 km² watershed located in northwest California and southwest Oregon (Figure 1). The Smith River is renowned for exceptionally clear water and large fluctuations in winter flow. The predominant rock material of the basin is from the hard, largely serpentine Josephine ophiolite (Stone 1993). Hard rock affords the basin an exceptionally high runoff rate (70%) and low levels of suspended sediment (EDAW 1980). Average annual rainfall in the basin is about 275 cm (EDAW 1980).

The Smith River basin includes over 100 km of large stream and fourteen tributaries known to be important in the production of chinook salmon and steelhead trout (Figure 1). The Smith River and all its tributaries are designated as both federal and California state Wild and Scenic Rivers, and the 95% of the basin that is under stewardship of the U.S. Forest Service is managed as the Smith River National Recreation Area (USDA 1995) and a key watershed (FEMAT 1993).

I chose my study segment from the lower Middle Fork, a sixth order stream that drains 803 km². The study segment was chosen because it was the largest river segment available with good access and depths not in excess of underwater visibility. The eight km segment, referred to hereafter as the Middle Fork, flows from confluence with the North Fork to a point five km above confluence with the South Fork where the confined and steep Oregon Hole Gorge begins. The study segment was moderately confined with

a gradient of 0.4%. Average flow for the summers of study (June 1 - August 3, 1993 and 1994) was 8.5 m³/s (300 ft³/s), average width 27 meters, and maximum depth 9.5 meters. Visibility in the Middle Fork, measured as the range at which a snorkeler could see five cm fish, was 7-10 m during summer.

Chinook salmon and steelhead trout were the most abundant fishes in the Middle Fork. Other fishes present were coastal cutthroat trout (*Oncorhynchus clarki clarki*), coastrange sculpin (*Cottus aleuticus*), prickly sculpin (*Cottus asper*), Klamath smallscale sucker (*Catostomus rimiculus*), and Pacific lamprey (*Lampetra tridentata*). Green sturgeon (*Acipenser medirostris*) was not observed, but was reported seen in the Middle Fork in 1973 (Jim Csutorus, Cresecent City, California).

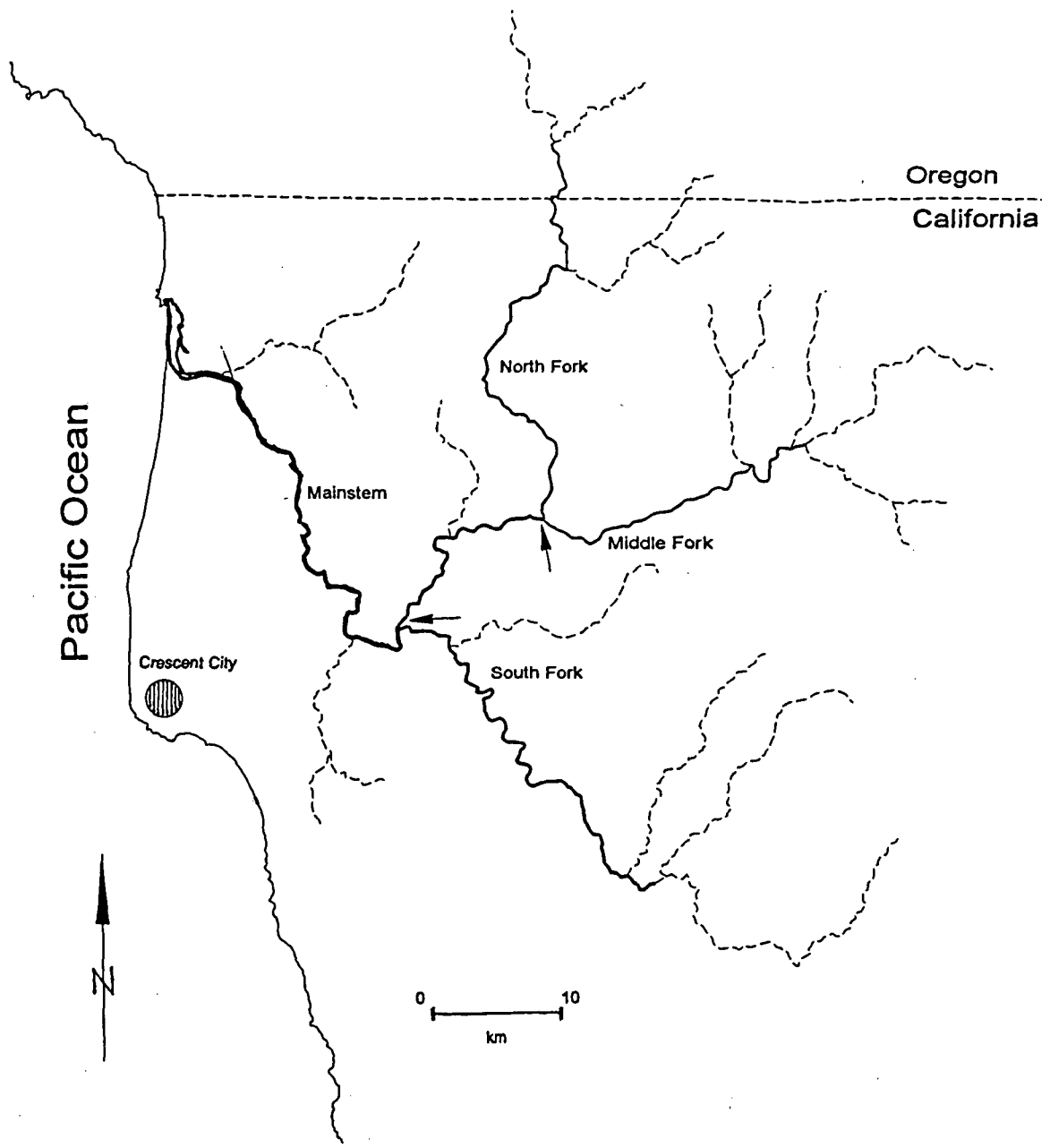


Figure 1. Geographical location, major branches, and major anadromous fish bearing tributaries (dashed lines) of the Smith River, California. Arrows indicate emigrant trapping sites which are located at either end of the study segment. Reproduced with permission from Six Rivers National Forest map.

METHODS

Target fishes of this study were underyearling chinook, and yearling or older juvenile steelhead (steelhead >1). The steelhead >1 category included fish from multiple age classes, indistinguishable by size (Peven et al. 1992). Underyearling steelhead were not counted because some were still emerging in early summer, and snorkel surveys are inappropriate for newly emerged fish which inhabit extremely shallow edgewater (Heggenes et al. 1990). Additionally, I felt that simultaneous enumeration of three types of fish would be so difficult as to introduce significant error.

Sampling Strategy

Stratification

I identified nine habitat types in the Middle Fork in 1993, and stratified accordingly; the entire study segment was divided into habitat units of nine types or strata. One backwater pool, and one side-channel were identified as unique habitats and considered two strata of one unit each. Fish abundance was enumerated in each of these unique habitats. Habitat units were sampled from the seven other strata. In 1994, the stratification was modified to accommodate an additional habitat type (in-pool run).

The habitat units of this study are the primary sampling units, but are located within or subsidiary to channel geomorphic units (channel units). This single stage

sampling design required, thus, two stages of stratification. First, channel units were classified according to the hydraulic characteristics outlined by Hawkins et al. (1993). Fast water units were classified as riffles if turbulent, and runs if not. Slow water units were classified as pools. Dam pools did not occur. The three channel units used occupy level II positions in the channel unit similarity dendrogram of Hawkins et al. (Figure 2). In a second phase, habitat units were classified from within each channel unit (Figure 3).

Channel units were classified on June 15, 17, and 22, 1993. A pass through the entire study segment was made on each date using a kayak. Notes on channel morphology with specific reference to depth, channel width, slope, turbulence, and velocity were taken on water resistant maps traced from aerial photographs. Relative turbulence and velocity were used to differentiate channel units. Multiple passes were performed to assure consistent classification and to observe changes in habitat characteristics over a short period of decreasing flow. From June 15 to June 22, 1993 mainstem Smith River flow decreased 36%, from 77.5 m³/s to 49.3 m³/s. Because sampling would take place at even lower flows, trend information was used in classifying channel unit types. For example, a long turbulent fast water unit began to show laminar flow through an intermediate area on June 22. The riffle was, therefore, broken into riffle-run-riffle in anticipation of the run fully developing. Several similar situations also occurred in recognizing smaller pools as they emerged from runs.

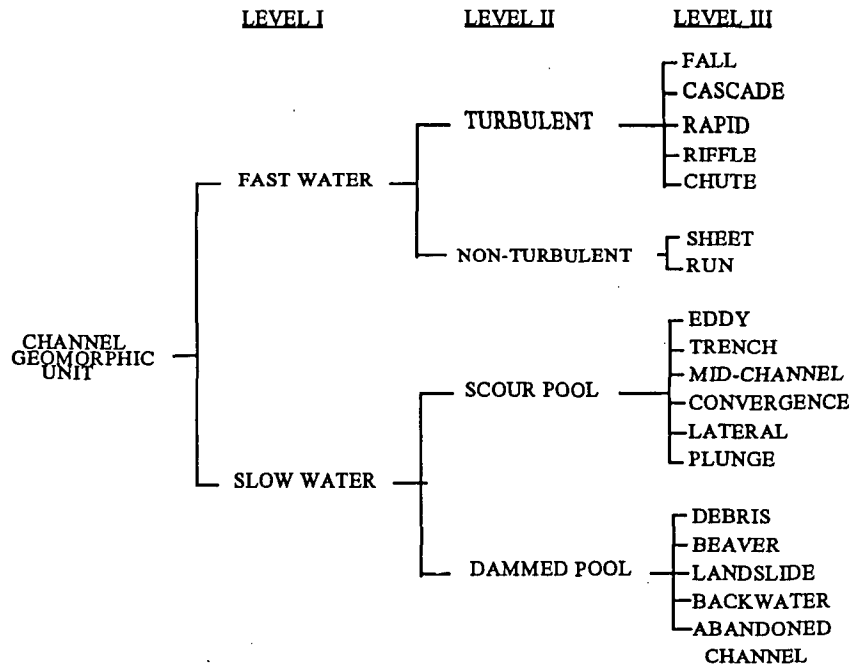


Figure 2. Channel unit similarity dendrogram from Hawkins et al. (1993) showing three levels of stratification.

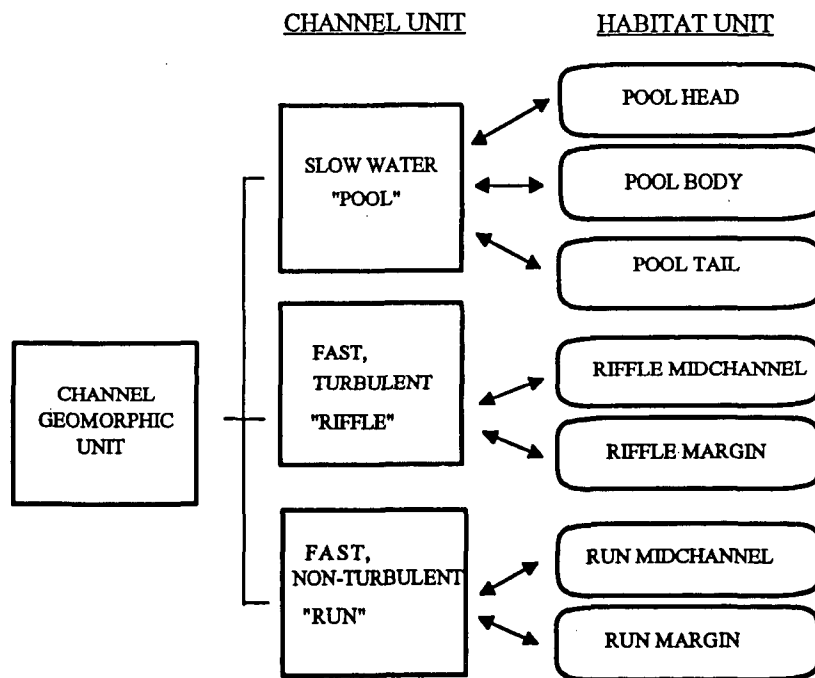


Figure 3. Habitat classification schematic used in the Middle Fork Smith River, California. Arrows indicate channel units are partitioned into habitat units.

Habitat units were classified within each channel unit according to the encompassing channel unit type and the hydraulic characteristics of turbulence, velocity and depth (Figure 4). Habitat types are defined as follows:

Pool Head - turbulent or relatively fast water (0.1 - 1.3 m/s) of the upstream portion of a pool. A pool head is a transition area from a fast water unit to a pool. From the fastwater unit/pool boundary, where water surface slope decreases and bed slope increases, the pool head extends downstream to where neither velocity nor turbulence remain conspicuous.

Pool Body - very slow (0.0 - 0.3 m/s), non-turbulent water within a pool.

Pool Tail - an area of increasing water velocity (0.1 - 1.0 m/s) at the downstream end of a pool. A pool tail is the transition of a pool into a riffle, and occurs where the wetted channel meets an alternate bar. The pool tail decreases in depth moving towards the downstream boundary of the pool where water surface slope increases sharply.

Riffle Margin - relatively slow water (0.0 - 0.5 m/s) of a riffle, found along each bank. Variable width and velocities of the margin result from local hydraulic-bank interactions.

Riffle Midchannel - the remainder of a riffle unit characterized by swift (0.5 - 2.0 m/s), turbulent water in or near the thalweg.

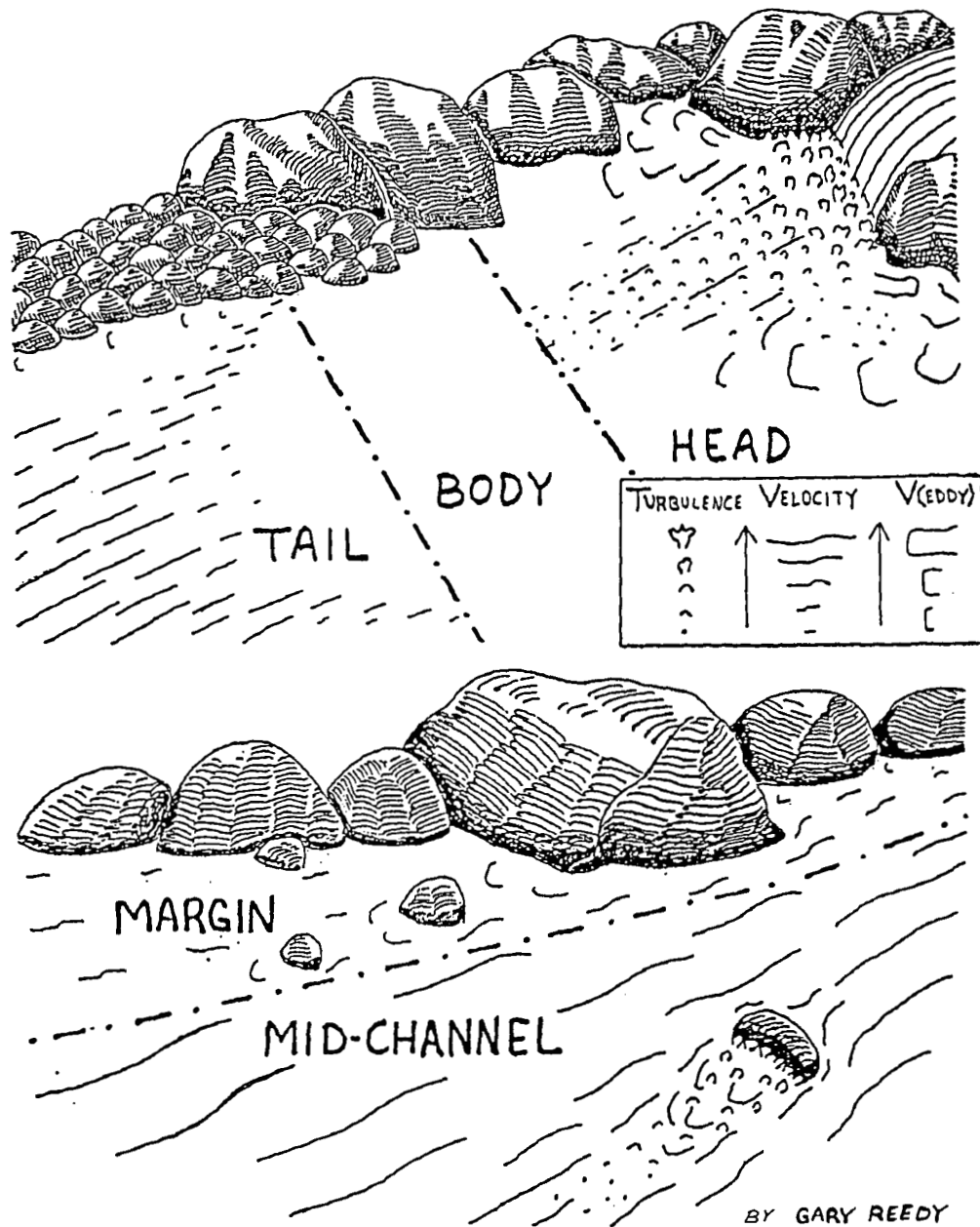


Figure 4. A pool and a fastwater channel unit (riffle or run) showing the subsidiary position of habitat units. Relative velocity and turbulence of habitat types can be interpreted using the symbol gradients in legend. $V(\text{eddy})$ is velocity in the upstream direction.

Run Margin - relatively slow (0.0 - 0.5 m/s), bank-associated water of a run.

Run Midchannel - the remainder of a run unit characterized by

swift (0.5 - 2.0 m/s), non-turbulent water in or near the thalweg.

In-pool Run - relatively swift (0.1 - 0.7 m/s), non-turbulent water of a pool

resulting from a channel-wide deposition which is not large enough to form a riffle and thereby does not separate the pool into two distinct channel units.

Depth and velocity were measured along a longitudinal transect of several pools to characterize pool head, pool body, and pool tail habitats (subsidiary pool habitats). I expected to classify one pool head, one pool body, and one pool tail for each pool; one riffle midchannel and two riffle margins for each riffle; and one run midchannel and two run margins for each run. Habitat classification deviated from this expectation due to the following conditions:

1. A pool may comprise a pool head flowing directly into a pool tail. No intermediate area of tranquil water in a pool resulted in no pool body in two of 24 pools.
2. A pool may flow into a run without the decreasing depth characteristic of a pool tail. Such chute-like transitions resulted in no pool tail in three of 24 pools, one of which had no pool body due to condition (1).

3. In long, trench shaped pools, depositional areas may occur that only mildly affect hydraulics at relatively low flow. These short, subsurface bars form natural breaks in very long stretches of slow water, resulting in multiple consecutive pool bodies. Five such areas were identified. These were classified as an additional habitat type, in-pool run, in 1994.
4. One riffle was split by an island, resulting in four margins. The split midchannel was treated as one subunit.

I assumed that the position, types and areas of habitat did not change between August 1993 and the period of equivalent flow in 1994. Habitat classification was therefore not repeated in 1994. Peak flow for the Mainstem Smith River during the winter of 1993/1994 was 65% (1104 m³/s) of bank-full. Bank-full was estimated at approximately 1700 m³/s using an annual maximum 1.5 year recurrence (Leopold et al. 1964) over 63 years, and represents the flow associated with channel shapes and dimensions (Wolman and Miller 1960).

A map of all habitat units was constructed using 1:5000 scale aerial photographs taken in August 1993. This map was used to modify the stratification for 1994. Three in-pool runs were delineated from portions of adjacent pool bodies and two riffles were changed to in-pool runs. The affects of these sampling frame adjustments on habitat area and numbers of units per stratum were carefully noted.

Sampling Within Strata

Fish abundance was sampled twice in 1993 and three times in 1994. The sampling occasions are denoted '93.1, '93.2, '94.1, '94.2, and '94.3, in chronological order, and comprised the following dates:

'93.1) 6/24/93 - 7/9/93

'93.2) 8/3/93 - 8/12/93

'94.1) 6/20/94 - 6/29/94

'94.2) 7/19/94 - 7/27/94

'94.3) 8/23/94 - 8/31/94

The five occasions range from nine to twelve days. The first and second sampling occasions in 1994 were chosen to correspond with flows during respective sampling occasions in 1993. A third, late summer sampling occasion was chosen for 1994 only.

Habitat units were systematically selected from each of the seven strata prior to '93.1. Sample sizes within strata ranged from seven to twelve. I assumed that stratum abundance would correlate with stratum variance, and sought to increase overall precision by adjusting sampling effort among strata according to expected abundances of fish among strata. In hindsight, choosing disproportionate sample effort among strata could have been soundly based only on differences among estimated stratum variances or costs of sampling within strata.

The following description of the sample selection process is stratum specific. Units were numbered consecutively from top to bottom of study segment. The first

sampling unit was selected from a random draw of the numbers 1 through k , the total number of units (N) divided by the sample size (n). Every k th unit thereafter was selected until n units were obtained. If N/n was not an integer result, then circular systematic selection was applied (Konijn 1973). In these cases, the habitats were mapped in a circle, with the last unit of the segment leading to the first again. A start unit was drawn at random from 1 through N . Every k th unit (k being the nearest integer to N/n) was then drawn until n units were obtained.

Samples were retained from one occasion to the next in order to decrease the variance of the estimated difference in abundance between two occasions (Raj 1968). Samples changed somewhat between years, however, as sample sizes were standardized at ten from each strata in 1994. Previously selected units were randomly discarded from too large a sample or previously unselected units were randomly added to too small a sample using a random number table in Dowdy and Wearden (1991).

In 1994, the sampling design was modified with respect to pools; fish were enumerated within all subsidiary habitats of ten selected pools. The modification was made to analyze relationships among subsidiary pool habitats and more adequately assess the utility of their stratification. The ten pools selected for 1994 were from the previously and systematically selected pool head sample. Two of twelve pools were removed from the sample following the procedure described above.

The method by which snorkelers visited selected habitat units for fish enumeration during each sampling occasion was considered important for minimizing

cost and co-dependency of observations. In 1993, a team of three snorkelers swam downstream through the study segment, stopping to enumerate fish in all selected subsidiary pool habitats. Afterwards, a team of two snorkelers made another downstream progression through the study segment to enumerate fish in all selected margin and midchannel units. Out of concern for co-dependency of observations, whereby a downstream progressing concentration or vacancy of fish could affect many counts, a slightly different pattern of sampling was used in 1994. The segment was divided into three reaches. Snorkeler teams still swam downstream from unit to unit, but began with the lowest reach and progressed upstream.

Sampling Within Habitats (Counting Fish)

The strategy used to enumerate fish in a habitat unit differed among strata. All but margin type subunits were subdivided into lanes that ran parallel to the riverbank. Lanes varied in width, but never exceeded the range of vision of snorkelers. Margins never exceeded 7 m in width, permitting one snorkeler to view the entire unit in one pass. One or two snorkelers were assigned to each lane or margin. When two snorkelers shared a lane or margin, they moved as close as possible and did not communicate, each counting all observable fish in the lane to generate paired independent counts.

Each habitat unit was considered separately in the process of subdivision. Lane boundaries in subsidiary pool units were a bank, the swimming path of an adjacent snorkeler, or infrequently an imaginary line connecting conspicuous boulders. The

number of lanes, their boundaries, and the snorkeler assignments were determined together. The process of subdivision sought to: (1) clearly delineate lane boundaries, (2) position snorkelers looking out from shaded areas rather than looking into shaded areas from sunlit areas, and (3) minimize the chances that a diver would have difficulty swimming or be forced to take a unintended course due to strong current.

To the following extent, subdivision was stratum specific:

Pool Heads. Snorkelers simultaneously swam two or three lanes in the upstream direction with the exception of one unit, which had four lanes, and one unit which had six lanes. The additional lanes were isolated by the thalweg.

Snorkelers swam them immediately following other lanes.

Pool Bodies. Snorkelers simultaneously swam two to three lanes in the upstream or in the downstream direction.

Pool Tails. Snorkelers simultaneously swam two to three lanes in the downstream direction. Only one unit was narrow enough for two lanes.

Margins. One or two snorkelers moved upstream through each unit. Snorkelers usually moved along a line near the bank where the water was slower and shallower, permitting the use of hands along the substrate. From this position, most fish were located between the snorkeler and the margin/midchannel boundary. The margin/midchannel boundary was

tentatively drawn before starting upstream, and was sometimes reinterpreted during the pass after closer underwater observation of local hydraulics.

Midchannels. In each of two to four downstream passes a snorkeler counted fish in a 2.5 - 4.0 m wide lane. The first two lanes were viewed simultaneously, and positioned so as not to overlap or include margin habitat.

All fish counting took place between 9:00 a.m. and 4:00 p.m. Snorkelers recorded lane counts on personal writing slates. Snorkelers wore fins, but no weight belts. Diving underwater to examine the depths or cover spots was not conducted because all target fish were assumed to be visible from the surface. Diving would have compromised the consistency of counts and was observed to cause a fright reaction in some fish. Snorkelers had previous experience identifying juvenile salmonids in their natural habitat, and practiced as a team before making actual counts.

All steelhead trout >80 mm and <250 mm were assumed to be steelhead >1 . The lower limit for steelhead >1 was assumed to be 90 mm on the second sampling occasion of each year, and 100 mm on the third. These assumptions regarding the shifting size separation of steelhead <1 and steelhead >1 were based on observation, and tested using the size frequency distribution of steelhead captured in fyke nets.

Habitat Area

Habitat unit areas were determined for potential use in fish abundance estimation and for estimation of observable fish densities during '93.2 and '94.2. Habitat dimensions were measured from aerial photographs assumed to accurately represent habitats during '93.2 and '94.2 when flow was similar to the day of photography.

Color, stereoscopic aerial photographs were taken from a helicopter on August 17, 1993 by Richard B. Davis Inc. of Smith River, California. Photo scale ranged from 1:5532 at the base of the study segment to 1:4836 at the top of the segment. Scale was calibrated using the Global Positioning System, and a different mean scale (1:5304, 1:5100, or 1:4836) was used for all area calculations in each of three stratified reaches.

Under 10x stereoscopic binoculars, all but margin/midchannel boundaries were discerned and drawn onto mylar sheeted prints. Streambanks, gravel bars, and habitat boundaries were digitized and the area of each subsidiary-pool, riffle, and run habitat calculated using the Design Cad program. Lengths of these units were measured from the digitized map, and widths calculated.

Visual estimates of mean width were made for the selected sample of margin and midchannel habitats during '93.2 and '94.2 and immediately prior to counting fish in each unit. Width estimates were multiplied by the appropriate run or riffle lengths to obtain margin and midchannel habitat areas. Margin area estimates were considered more accurate than midchannel area estimates, due to their larger sample and smaller widths. The average proportion of margin areas to their respective riffles or runs was used,

therefore, in calculating total area for both margin and midchannel habitats. Simple summation was used for the total area of all other habitat types.

Estimating Fish Abundance

This sampling design generated independent estimates of observable fish abundance in seven strata using simple expansion of stratum means (mean per unit estimation). The modified design of 1994 generated estimates in eight strata due to the addition of in-pool runs. Subsidiary pool habitats were not independent in 1994 due to their grouping within ten sampled pools.

A ratio approach to abundance estimation was tested for August 1993 and July 1994 using habitat area as an auxiliary variable. To increase sampling precision, ratio estimation was conducted when the correlation between fish abundance and habitat area was greater than one-half the ratio of the coefficient of variation for the auxiliary variable (area) to the coefficient of variation for the target variable (abundance)(Cochran 1977). The performance of ratio estimation was compared to mean per unit estimation in these cases.

The following notation is stratum-specific:

y_i = number of observable fish in unit i ; $i = 1, 2, \dots, N$

y_{ij} = number of observable fish in lane j of unit i ; $j = 1, 2, \dots, M$

N = total number of units in stratum

n = number of units wherein fish were counted

M_i = total number of lanes in unit i

m_i = number of lanes in unit i wherein fish were counted

x_i = area of unit i (m^2)

X = total habitat area (m^2)

Summations are over the units in which fish counts are made, unless otherwise noted.

With the exception of riffle midchannels and run midchannels, all lanes of a selected unit were sampled (i.e., $m_i = M_i$). The above notation is consistent with Hankin and Reeves (1988). The formulas are found in Cochran (1977), unless otherwise noted.

Mean Per Unit Type Estimators

The sample mean was estimated as:

$$(1) \quad \bar{y} = \frac{\sum_{i=1}^n y_i}{n}.$$

Variance of the sample mean was estimated as:

$$(2) \quad \hat{V}(\bar{y}) = \frac{(N-n)}{Nn} s^2$$

where $s^2 = \frac{\sum_{i=1}^n (y_i - \bar{y})^2}{n-1}$.

Total observable fish was estimated as:

$$(3) \quad \hat{Y} = N\bar{y}.$$

Variance of the estimated total observable fish was estimated as:

$$(4) \quad \hat{V}(\hat{Y}) = N^2 \hat{V}(\bar{y}).$$

Ratio Estimators

The ratio of fish abundance to habitat area (i.e., density) was estimated as:

$$(5) \quad \hat{R} = \frac{\sum^n y_i}{\sum^n x_i}$$

The ratio is not an unbiased estimate of the true ratio of fish abundance to habitat area (Cochran 1977). The bias component to mean square error is small, however, relative to the variance component considered here.

Variance of the estimated ratio was estimated as:

$$(6) \quad \hat{V}(\hat{R}) = \frac{(N-n) \sum^n (y_i - \hat{R}x_i)^2}{Nn\bar{X}^2 (n-1)}$$

where $\bar{X} = \frac{X}{N}$ (Raj 1968).

Total observable fish was estimated as:

$$(7) \quad \hat{Y}_R = X\hat{R}$$

Variance of the estimated total observable fish was estimated as:

$$(8) \quad \hat{V}(\hat{Y}_R) = X^2 \hat{V}(\hat{R})$$

Two Stage Estimation For Midchannel Type Units

Fast water in midchannel habitats prohibited the synchronous snorkeling of all lanes, requiring subsampling of two or more lanes. The number of fish in midchannel units (y_i) was estimated using a mean per subunit (lane) estimator:

$$(9) \quad \hat{y}_i = M_i \frac{\sum_{j=1}^m y_{ij}}{m} = M_i \bar{y}.$$

Variance of the estimated number of fish in a unit was estimated as:

$$(10) \quad \hat{V}(\hat{y}_i) = M_i^2 \frac{(M-m)}{Mm} \frac{\sum_{j=1}^m (y_{ij} - \bar{y}_{ij})^2}{m-1}.$$

Total observable fish was estimated as:

$$(11) \quad \hat{Y} = N \sum \frac{\hat{y}_i}{n}.$$

Variance of the total number of fish resulted from a two-stage variance estimator,

$$(12) \quad \hat{V}(\hat{Y}) = N^2 \frac{N-n}{N} \frac{s_1^2}{n} + \frac{N}{n} \sum_i M_i^2 \left(\frac{M_i - m_i}{M_i} \right) \frac{s_{2i}^2}{m_i}$$

$$\text{where } s_{2i}^2 = \frac{\sum_{j=1}^m (y_{ij} - \bar{y}_{ij})^2}{m-1}.$$

Assumptions And Limitations Regarding Bias

Targeting observable fish, rather than the actual number of fish, does not entirely manage the problem of bias unless one is willing to assume constant proportional bias (i.e. constant percent underestimation) over all observations. I believe that depth, velocity and substrate affect bias, but that since these attributes are indirectly stratified, it may be reasonable to assume constant proportional bias within, but not among strata. Although independent estimates of stratum totals and variances of stratum totals are simply additive in the generation of estimates over all strata (Raj 1968), such estimates were not generated in this thesis. When proportional bias varies among strata, the estimate over all strata is confounded. If distribution of fish among strata changes over time, comparisons of the estimate over all strata will err.

With the exception of pools in 1994, this design did not estimate abundance for individual channel units. Total abundance for channel unit types was estimated as the sum of estimates for subsidiary habitat types under the assumption that proportional bias for those habitats was similar. Channel unit estimates are not used, however, in temporal analyses.

Change In Abundance

For statistical inference of change in abundance, the following comparisons were made: '93.1 to '93.2, '94.1 to '94.2, and '94.2 to '94.3. In addition, interannual change in abundance was examined by comparing the average of 1993 sampling periods to the

average of '94.1 and '94.2. Comparisons were made by estimating the difference in total fish abundance between two occasion, or two years, for each habitat type. Estimated differences (changes) were bound in confidence intervals. To maintain an experiment-wise type 1 error rate (α) of 0.05, the Bonferroni technique was used to establish a comparison-wise error rate (α') of 0.0125 (Sokal and Rohlf 1995). Confidence intervals excluding zero indicated statistically significant change in abundance. The formulas used for this approach are found in Raj (1968).

The habitat-specific difference in total abundance between sampling periods was estimated as:

$$(13) \quad \hat{d} = \hat{Y}_2 - \hat{Y}_1.$$

For common samples, variance of the difference was estimated as:

$$(14) \quad \hat{V}(\hat{d}) = \hat{V}(\hat{Y}_1) - 2\hat{COV}(\hat{Y}_1, \hat{Y}_2) + \hat{V}(\hat{Y}_2)$$

where $\hat{COV}(\hat{Y}_1, \hat{Y}_2) = \text{covariance of estimated totals from period 1 and 2}$
 $= \hat{COV}(N\bar{y}_1, N\bar{y}_2) = N^2 \hat{COV}(\bar{y}_1, \bar{y}_2).$

Covariance for two random variables is calculated using:

$$(15) \quad \hat{COV}(\bar{x}, \bar{y}) = \frac{N-n}{Nn} \frac{\sum (x_i - \bar{x})(y_i - \bar{y})}{n-1} \quad (\text{Cochran 1977, eqn. 2.15}).$$

If samples from two occasions had been drawn independently, covariance was assumed to equal to zero.

Within a year, k, the average over two sampling occasions was estimated as:

$$(16) \quad \bar{Y}_k = 1/2 (\hat{Y}_1 + \hat{Y}_2).$$

Variance of the average over two sampling occasions was estimated as:

$$(17) \quad \hat{V}(\bar{Y}_k) = 1/4 [\hat{V}(\hat{Y}_1) + 2\hat{COV}(\hat{Y}_1, \hat{Y}_2) + \hat{V}(\hat{Y}_2)].$$

The difference between yearly averages is obtained by following the above calculations with Equations 13 and 14.

Assessing Measurement Error With Paired And Repeat Counts

Sampling error and measurement error both contribute to the total errors of estimation. The variance estimators, however, express sampling error only.

Measurement errors, which arise from several sources and result in inaccurate enumeration of observable fish, need to be assessed in order to evaluate the legitimacy of this sampling design. I evaluated the utility of visually estimating fish abundance in the Middle Fork by comparing the magnitude of measurement error to that found in snorkel surveys of other streams. I also examined the contribution to measurement error from different sources in order to identify limitations or needs for improved sampling procedures.

Measurement error refers to the difference between the observed and actual numbers observable fish in a unit, and is indicated by estimates of repeatability.

Variability in paired and repeat counts were thus used to indicate measurement error.

Paired counts were made simultaneously by two observers moving through a unit together. To maintain independence of counts, paired observers attempted not to

obstruct each others view, and did not communicate during counting. Alternatively, repeats were a pair of counts in the same unit made ten minutes to four hours apart. Repeats in 1994 were made by two different observers, or by entirely different assignment of observers to lanes. Each observer recorded counts separately. Coefficient of variation ($CV = \text{standard deviation}/\text{mean}$) was calculated for each paired count or repeat as a standardized measure of repeatability (measurement error).

The expected sources of measurement error in this study were observer variance, fish displacement, and natural changes in the numbers of fish over the sampling period. Observer variance results from the variable performance of snorkelers in counting moving fish, and was assessed by comparing CVs among species, habitats and occasions that differed in abundance. The other two sources were evaluated in several ways.

Fish movement resulting from the activity of snorkelers is termed fish displacement, and was assessed by testing repeats for a lower second count. Additionally, two tests were designed to determine if one snorkeling protocol displaced fish more than an alternative protocol. On one sampling occasion, snorkelers swam some pool bodies upstream first, then downstream. Snorkelers swam an equal number of randomly selected pool bodies vice versa. A paired T-test was used to determine if downstream swimming snorkelers counted fewer fish than upstream swimming snorkelers. In another test, snorkelers swam, and then repeated, some pool tails after no prior disturbance to the adjacent pool body. Snorkelers swam the second pass of an equal number pool tails after a 15 minute wait period. A paired T-test was used to see if pool

tail counts made immediately after pool body counts had higher numbers of fish than pool tail counts made without prior "sweeping" of the pool body.

Natural movement by fish through and between units causes temporal variation in fish abundance within a sampling period. Because most movements of juvenile steelhead and chinook occur overnight (USFWS 1995), I predict that repeats consisting of counts separated by one or more days (revisits) will have higher variance than repeats consisting of counts made within the same day. Furthermore, I estimated a mean abundance for individual units by averaging repeats, and predicted that if overnight movements occurred, the inclusion of a revisit count in that estimate would 1) change the estimated mean abundance, and 2) increase the variance of the estimated mean. The degree of such affects may be in proportion to the amount of movement by fish.

Repeats which included a zero fish sighting were excluded from analyses because they tended to have CVs outlying the other values. Repeats were not conducted on midchannel habitats because variability in repeat counts could not be separated from secondary sampling error.

Emigrant Trapping

Fyke-nets were used at two Middle Fork sites (Figure 2) located 12 km apart. Captured juvenile chinook and steelhead >1 were assumed downstream migrants (emigrants). Numbers captured were used to plot trends in emigration. Fork length measurements (nearest mm) were used to generate size frequency distributions, and to

make emigrant size comparisons between trap sites. Fish caught at the upper site (Site 0 km) were marked on three consecutive days in hopes that recaptures at the lower site (Site 12 km) might allow estimation of growth and migration rates. Marked fish were also released at Site 12 km. The assumption regarding capture of downstream migrants and not locally residing fish was thereby tested for each trap site.

The fyke-nets were constructed of 8 mm nylon mesh, opened at the mouth 1.22 m (4 ft) square, contained three sets of internal walls to prevent escape of fish, and were attached to a 0.5 x 0.7 x 1.2 m wooden live box (Figure 5). Traps were positioned in the thalweg of pool heads using ropes, pulleys, and a fixed anchor on each bank. Traps were set one-half to two hours before sunset and retrieved at least one hour after sunrise the following morning. The daytime period (9 am to 7 pm) was also fished on three consecutive occasions to test the assumption of negligible daytime migration.

Site 12 km was fished more intensively in 1994 to generate index estimates of total emigrants and to plot trends in emigration. Traps were operated on 42 nights between June 27 and August 30. Two fyke-nets strapped together side-to-side were used on most occasions.



Figure 5. Fyke-net used in the Middle Fork Smith River, California with live-box (1.2 m long) attached.

Detailed measurements of flow into the nets was taken before retrieval. Number of emigrants past Site 12 km on a given night was estimated using the following equation:

$$(18) \quad \hat{E} = e \frac{Q}{q}$$

where e = the number of fish caught

q = flow into the net

Q = flow in the Middle Fork.

Flow into the net was calculated by dividing the fyke-net opening into three vertical panels. Average velocity was measured for each panel, and the product of velocity and area summed per panel. Average velocity for each panel was calculated as the average of readings taken at 0.2 x depth (d) and 0.8 d positions. Velocity readings were taken over twenty second intervals with a Marsh-McBirney Model 201 flowmeter. Depth of the panels was the distance from the bottom of the fyke-net mouth to the water surface. The bottom edge of the fyke-net mouth was always within 40 cm of the river bed. The position of the top of the fyke-net was always recorded as height above or below water surface.

Equation (18) estimated total emigrants using the proportion of flow through the net. Because downstream migrants did not distribute evenly with the flow (Mains and Smith 1964), this estimator merely generated an index for total emigrants that was more useful in monitoring trends in emigration than raw catch numbers. The index adjusted catch numbers to account for variably placed nets and the increasing proportion of flow captured by the nets over the course of summer.

Flow in the Middle Fork was calculated as a constant proportion of the flow in the mainstem Smith River which is continuously available from an automated gage located in Jedidiah Smith State Park. Middle Fork flow was measured and calculated at two cross-sections on July 14, 1994, using thirty panels. A proportional constant was calculated as the average estimated flow divided by the existing mainstem flow.

To test the assumption that catches did not contain locally residing fish, several batches of fish were marked and released at the trap site where they had been captured. At Site 12 km, 30 chinook and 12 steelhead >1 captured on July 16, 1994 were marked with caudal fin clips. Thirty chinook and 12 steelhead >1 were again marked and released at Site 12 km on August 8, 1994. One hundred thirty chinook captured at Site 0 km on July 29-31, 1993 were marked with caudal fin clips and a blue stain to the caudal fin (administered with a Pan-Jet dye gun). Each site was surveyed by a snorkeler on days following release to determine the presence of marked fish that were not recaptured. In an effort to measure migration rate, Site 12 km was operated on 16 of the 19 nights following initial release of marked fish from Site 0 km..

Growth of fish during use of the Middle Fork was interpreted from the difference in length between fish caught at Site 0 km and fish caught at Site 12 km. Lengths were compared for each week when both traps were operated. Within such weeks, Site 0 km was fished 1-4 nights before Site 12 km to make potential estimates of growth more conservative.

RESULTS

Flow And Water Temperature

The average of two Middle Fork flow estimates (5.48 m³/s, 5.51 m³/s) was divided by the concurrent mainstem flow for a proportional constant of 0.40. All reported flow levels for the Middle Fork herein are calculated using 0.40 x estimated mainstem flow. An alternative technique using basin area would have generated very similar results. The Middle Fork contributes 41% of the gauged watershed area.

Flow decreased throughout each sampling occasion and between sampling occasions of each year (Table 1), except for 0.5 cm of rain which caused a 2% increase in flow on July 22, 1994. Flow was greater on all summer dates in 1993 compared to respective dates in 1994 (Figure 6).

Water temperature was taken with hand-held thermometers on the morning of each trap occasion and at the beginning and end of each snorkeling session. Water temperature varied among sampling occasions, and between years (Table 1). On most days, solar radiation increased water temperature 1-3 °C from morning to late afternoon.

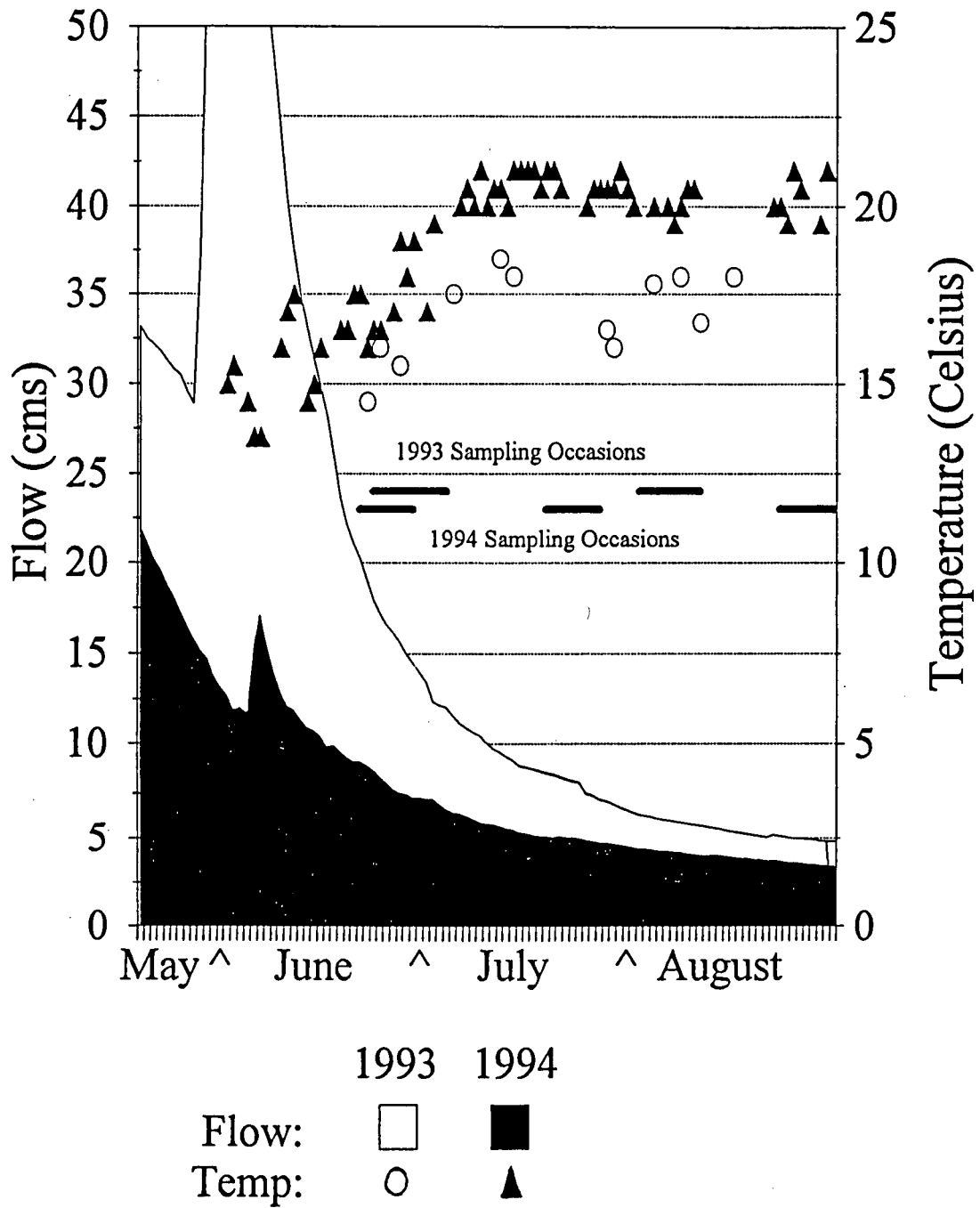


Figure 6. Summer hydrographs (May 20 to August 31, 1993 and 1994) for the Middle Fork Smith River, California with morning temperature readings (taken between 0700 and 1000 hr.. at Site 12 km) and sampling occasions (horizontal bars) superimposed. Arrows indicate the first of each summer month. The 1993 storm event peaked at 169 m³/s (6000 cfs) on June 1.

Table 1. Maximum and minimum recorded flow levels and water temperatures in the Middle Fork Smith River, California for all sampling occasions. Temperature readings were pooled per sampling occasion.

Sampling Occasion	Flow (m ³ /s)		Temperature (°C)		
	Start	End	Maximum	Minimum	Mean
'93.1: 6/24/93 - 7/09/93	17.8	10.3	17.5	14.5	15.9
'93.2: 8/03/93 - 8/12/93	6.8	6.1	18.0	16.7	17.5
'94.1: 6/20/94 - 6/29/94	9.2	7.5	18.5	16.0	17.3
'94.2: 7/19/94 - 7/27/94	5.1	4.8	22.0	20.5	20.9
'94.3: 8/23/94 - 8/31/94	3.7	3.4	21.0	19.5	20.2

Habitat Inventory

I identified 26 pools, 46 riffles, and 26 runs on the Middle Fork in June 1993.

The upstream and downstream boundaries of these channel units corresponded with fixed geomorphic features and did not change under the range of low summer flow. The number of habitat units also remained unchanged through the summer of 1993, but boundaries between some habitat units shifted with decreasing flow (see Habitat Area). Thus, within the summer of 1993 the listing and physical layout of sampling units (sampling frame) did not change except for some area adjustments.

Prior to sampling in 1994, I made adjustments to the sampling frame in order to provide an improved stratification based on observations throughout the summer of 1993. The sampling frame changed in three minor ways: (1) 1993 classification errors

were corrected by deleting an exceptionally short run and converting a pool tail to an extended part of an existing run, (2) three in-pool runs were delineated from pool body/pool body transition areas as described under Methods, and (3) two pairs of pools were merged by changing the linking fastwater habitats to in-pool runs.

In order to apply the improved stratification to 1993 estimates and to allow appropriate interannual abundance comparisons, the original 1993 sampling frame was adjusted to match, as closely as possible, the 1994 sampling frame without affecting units sampled. Making improvements to the stratification used in 1994 without adjusting the 1993 sampling frame would have resulted in several problems. For example, estimates of fish abundance in pool heads in 1993 would have been artificially high relative to the estimates in 1994. Remaining disparities between the 1993 and 1994 frames are an additional pool tail, and the partial recognition of in-pool runs in 1993 (Table 2).

Habitat Area

Decreasing flows changed habitat areas as channel widths decreased and pool head/pool body boundaries gradually moved upstream. I observed pool tail lengths to remain constant. Margin widths varied in their response, but usually decreased. Midchannel widths appeared to change in proportion to channel width. Lengths of individual channel units did not change with flow.

Table 2. Number of units per habitat stratum (N_h) and stratum sample sizes (n_h) in the Middle Fork Smith River, California during summers of 1993 and 1994. Sampling frame adjustments are shown by the two N_h columns for 1993.

HABITAT	1993			1994	
	N_h	adjusted N_h	n_h	N_h	n_h
Pools	26	24	--	24	10
Pool Heads	26	24	12	24	10
Pool Bodies	27	27 ^a	7 ^a	27 ^b	12 ^b
Pool Tails	22	21	7	20	7
In-pool Runs	--	2	0	5	4
Riffles	46	45	--	45	--
Riffle Margins	94	92	10	92	10 ^d
Riffle Midchannels	46	45	7	45	10
Runs	26	24	--	24	-- ^c
Run Margins	52	48	10	48	10 ^c
Run Midchannels	26	24	7	24	10 ^c

a) Three sampled pool bodies include portions of three undesigned in-pool runs during 1993.

b) Only 11 pool bodies sampled from 26 in '94.1.

c) In lieu of non-differentiating subunits at low flow, 5 whole runs sampled in '94.3.

d) Only 9 visited in '94.1

Average flows during '93.2 and '94.2 were 18% greater and 8% less, respectively, than flow on August 17, 1993 ($5.4 \text{ m}^3/\text{s}$) when aerial photos were taken. Individual habitat unit areas were assumed equal among the two sampling occasions and the date of aerial photography (see Methods). Habitat dimensions were measured from digitized aerial photographs (Appendix A), and supplemented with field estimates of margin width to calculate habitat area for '93.2 and '94.2 sampling occasions (Table 3).

Sixty two percent of the total wetted area of the Middle Fork was classified as pool habitat (Table 3). Of the total pool area, 62% was classified as pool body habitat, 23% as pool head habitat, and 11% as pool tail habitat. Area was far more variable for pool body habitat units than for any other habitat type; the coefficient of variation for pool body area was 1.2 compared with 0.5 for the next most variable habitat types (riffle margins and pool heads). Dimensions and variation in size were similar for runs and riffles (Table 3). The proportion of fast water channel units classified as margin habitat, however, was greater for runs (52%) than riffles (29%). Run margins averaged 6.4 m in width compared to 3.3 m for riffle margins (Table 3).

Table 3. Habitat dimensions and areas for the Middle Fork Smith River, California in August, 1993 and 1994 with standard deviations in parentheses. Estimated values are accompanied by standard error in brackets. Margin and midchannel lengths were not measured (indicated by dashed lines), but for calculation of area, were assumed, on average, equal to run or riffle mean length. Failure of total area or % total area for subsidiary habitats to sum to respective channel unit values is due to rounding errors. Failure of total area and % total area for channel units to sum to overall total area and 100%, respectively, is also due to rounding errors. Two margins contribute to each riffle or run.

Habitat	Mean Length (m)	Mean Width (m)	Mean Area (m ²)	Total Area (m ²)	% Overall Total Area
Pool	203 (123)	29 (8)	6437 (5262)	154493	61.8
Pool Head	54 (19)	27 (7)	1443 (603)	34631	13.9
Pool Body	113 (94)	28 (8)	3535 (4083)	95436	38.2
Pool Tail	23 (14)	41 (23)	869 (447)	17377	7.0
In-pool Run	47 (18)	22 (36)	1027 (407)	5137	2.1
Riffle	55 (34)	23 (7)	1300 (882)	56925	22.8
Riffle Margin	-- --	3.3 ^a [0.5]	182 [30]	16698 [2732]	6.7
Riffle Midchannel	-- --	16.3 ^b NA	894 ^b NA	40227 ^b NA	16.2
Run	63 (33)	26 (11)	1534 (739)	36824	14.7
Run Margin	-- --	6.4 ^a [0.6]	403 [38]	19353 [1835]	7.7
Run Midchannel	-- --	12 ^b NA	728 ^b NA	17470 ^b NA	7.0
Overall Total Area:				248242	

a) Mean visual estimate of 10 margin widths.

b) Calculated by subtracting total margin area from total channel unit area.

Depth And Velocity Measurements

Depth and near surface velocity were measured along longitudinal transects in four pools in early August 1994 to characterize these attributes among subsidiary pool habitat types and to graphically represent the partitioning of pools that occurred (Figure 7). Pools were selected to represent the range of variation in pool morphology.

Range of measured depth and velocity were similar between pool heads and pool tails. Velocity was low for pool bodies and varied little despite changes in depth. In-pool runs were conspicuous zones of increased velocity within pool bodies. In a pool followed by a run instead of a riffle, velocity increased towards the end of the pool without any decrease in depth (Figure 7, B).

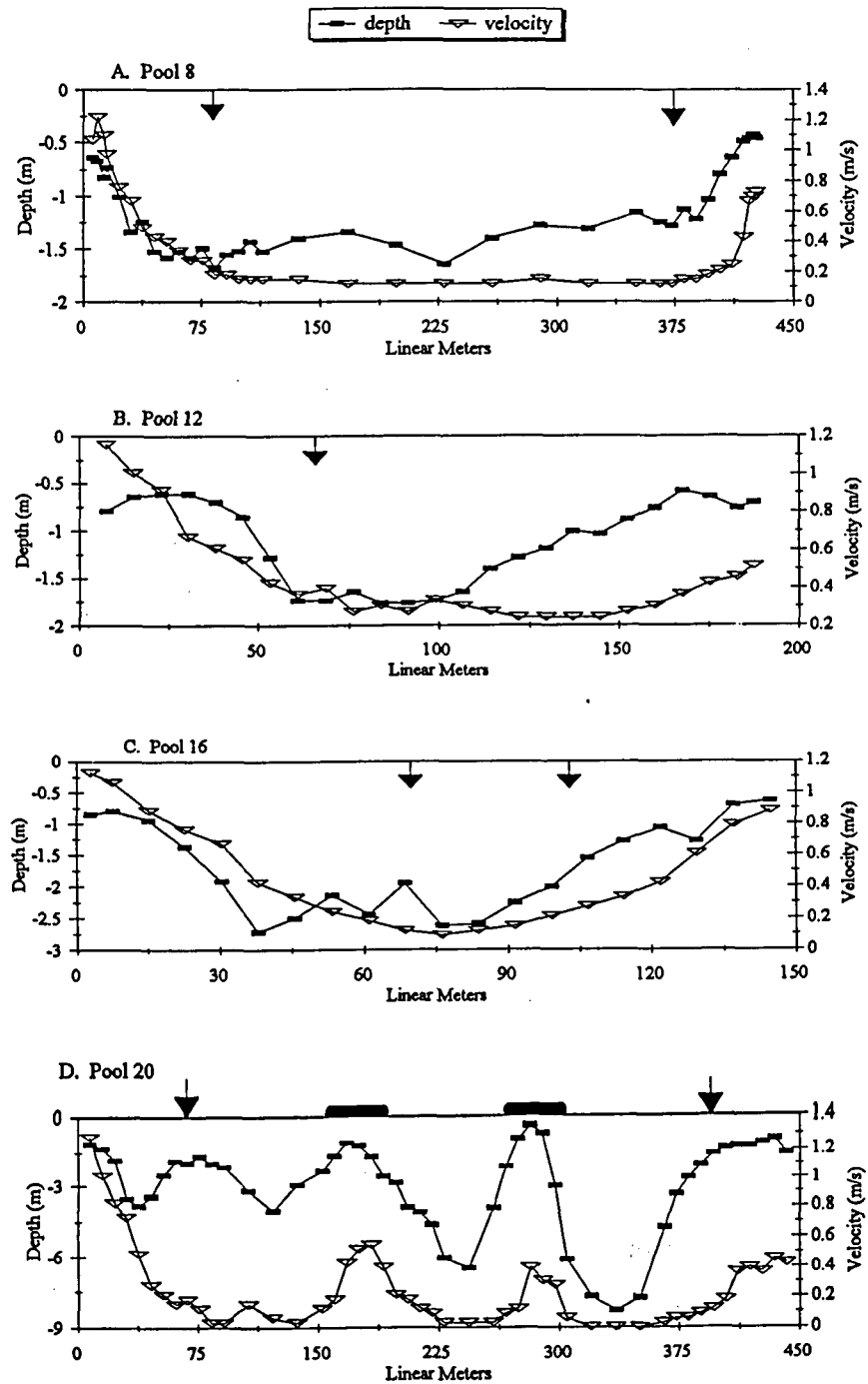


Figure 7. Depth and near-surface velocity on a longitudinal profile in four pools of the Middle Fork Smith River, California as measured August 2 - 4, 1994. Pool head/pool body and pool body/pool tail boundary locations from July 1994 are marked with an arrow. Pool 12 (B) was followed by a run and did not contain a pool tail. The horizontal bands in D (Pool 20) match the position of in-pool runs.

Estimated Fish Abundances

Visual estimates of fish per sample units (Appendix B) were used to estimate mean and total fish abundances with respective sampling variances (chinook, Table 4; steelhead >1, Table 5) in all habitat types at all sampling occasions using mean per unit estimators. Pools in all 1994 occasions and runs in '94.3 were sampled as whole units, but estimated totals for other channel unit types are the sum of estimates for subsidiary habitats. Estimated means for these channel units are the product of dividing totals by N_h (the number of total units of that channel type).

Ratio estimators were used to increase precision in fish abundance estimation during '93.2 and '94.2 when indicated by an adequately strong relationship between the target variable (fish abundance) and an auxiliary variable (habitat area) (Cochran 1977). For these data, the threshold for an adequately strong relationship was found to be equivalent to the $\alpha=0.05$ significance level for correlation between fish abundance and habitat area (Table 6). Accordingly, ratio estimation was used for steelhead >1 in pool body habitats and for both species in riffle margin habitats. In these cases, estimated sampling variance was lowered 56%, on average, and by more than 70% for pool body steelhead >1 (Table 7). Improvements in sampling precision afforded by habitat area relationships were used in graphical representation of chinook (Figure 8) and steelhead >1 (Figure 9) abundance, but not for estimated change in chinook (Figure 10) and steelhead >1 (Figure 11) abundance.

Table 4. Estimated mean (A) and total (B) observable chinook per habitat type in the Middle Fork Smith River, California with standard error (square root of estimated sampling variance) in parentheses. Dashed lines indicate data were not available for estimates.

A. Mean Chinook					
Habitat	'93.1	'93.2	'94.1	'94.2	'94.3
Pool	519 (60)	399 (67.4)	1078 (182.6)	474 (57.4)	63 (11.3)
Pool Head	279 (25.9)	202 (32.0)	491 (28.8)	280 (47.0)	39 (9.4)
Pool Body	192 (47.2)	105 (46.6)	458 (165.7)	117 (25.9)	14 (4.3)
Pool Tail	27 (9.7)	89 (31.7)	89 (19.9)	72 (18.4)	10 (4.8)
In-pool Run	- -	- -	81 (11.3)	15 (2.3)	2 (1.0)
Riffle	16 (6.5)	5 (2.3)	24 (7.7)	8 (3.0)	1 (0.8)
Riffle Margin	8 (3.2)	3 (1.4)	12 (3.8)	3 (1.4)	1 (0.4)
Riffle Midchan.	0 (0)	0 (0.1)	0 (0.2)	2 (1.0)	0 (0)
Run	52 (12.5)	17 (4.5)	72 (10.2)	24 (5.8)	9 (2.7)
Run Margin	23 (6.1)	6 (1.9)	29 (4.8)	8 (2.1)	- -
Run Midchan.	6 (2.4)	5 (2.5)	13 (3.4)	9 (4.0)	- -
B. Total Chinook					
Pool	12447 (1432)	9565 (1617)	25875 (4381)	11384 (1377)	1521 (271)
Pool Head	6698 (621)	4849 (768)	11775 (692)	6725 (1129)	927 (226)
Pool Body	5188 (1275)	2847 (1258)	11909 (4308)	3155 (698)	390 (115)
Pool Tail	561 (203)	1869 (665)	1784 (398)	1430 (367)	193 (97)
In-pool Run	- -	- -	407 (57)	74 (12)	11 (5.0)
Riffle	704 (292)	236 (106)	1070 (349)	357 (133)	55 (35)
Riffle Margin	704 (292)	230 (106)	1053 (349)	281 (124)	55 (35)
Riffle Midchan.	0 (0)	6 (5.9)	17 (7.6)	77 (47)	0 (0)
Run	1244 (299)	415 (109)	1725 (245)	572 (139)	206 (65)
Run Margin	1199 (294)	397 (91)	1409 (231)	360 (100)	- -
Run Midchan.	145 (57)	117 (60)	316 (82)	212 (97)	- -
Backwater	88 (0)	3 (0)	74 (0)	0 (0)	0 (0)
Side-channel	11 (0)	1 (0)	17 (0)	7 (0)	1 (0)

Table 5. Estimated mean (A) and total (B) observable steelhead >1 per habitat type in the Middle Fork Smith River, California with standard error (square root of estimated sampling variance) in parentheses. Dashed lines indicate data were not available for estimates.

A. Mean Steelhead >1					
Habitat	'93.1	'93.2	'94.1	'94.2	'94.3
Pool	167 (15.4)	337 (72.5)	729 (95.2)	831 (116.3)	642 (112.8)
Pool Head	104 (12.3)	139 (16.9)	270 (15.5)	383 (49.5)	301 (36.8)
Pool Body	21 (4.8)	99 (59.8)	203 (82.9)	74 (28.9)	18 (4.4)
Pool Tail	44 (8.5)	99 (24.1)	247 (32.9)	426 (120.0)	383 (127.8)
In-pool Run	- -	- -	156 (7.6)	51 (15.0)	2 (0.7)
Riffle	40 (6.0)	38 (8.4)	84 (14.2)	113 (23.2)	98 (23.5)
Riffle Margin	14 (2.7)	11 (2.6)	28 (6.2)	30 (7.7)	20 (6.8)
Riffle Midchan.	11 (2.2)	16 (6.5)	28 (6.2)	52 (17.0)	57 (18.9)
Run	61 (6.7)	63 (5.7)	167 (14.2)	139 (13.1)	131 (24.9)
Run Margin	19 (3.0)	16 (2.0)	46 (5.4)	22 (3.2)	- -
Run Midchan.	24 (3.0)	31 (4.0)	76 (9.3)	94 (11.4)	- -
B. Total Steelhead >1					
Pool	3999 (369)	8088 (1741)	17485 (2285)	19949 (2790)	15399 (2707)
Pool Head	2493 (296)	3327 (405)	6482 (373)	9183 (1189)	7229 (884)
Pool Body	579 (130)	2685 (1616)	5274 (2156)	1997 (779)	497 (119)
Pool Tail	927 (179)	2076 (506)	4947 (657)	8513 (2400)	7662 (2556)
In-pool Run	- -	- -	781 (38)	256 (75)	11 (3.7)
Riffle	1790 (269)	1713 (378)	3785 (637)	5081 (1045)	4424 (1056)
Riffle Margin	1302 (250)	1012 (242)	2539 (573)	2728 (712)	1849 (623)
Riffle Midchan.	489 (101)	701 (290)	1246 (277)	2354 (765)	2575 (852)
Run	1466 (160)	1511 (136)	4016 (342)	3326 (315)	3144 (598)
Run Margin	888 (143)	756 (97)	2191 (260)	1063 (153)	- -
Run Midchan.	578 (73)	755 (96)	1825 (222)	2263 (275)	- -
Backwater	25 (0)	4 (0)	24 (0)	1 (0)	1 (0)
Side-channel	16 (0)	26 (0)	44 (0)	121 (0)	75 (0)

Table 6. Correlation coefficients (r) for fish abundance and habitat area in the Middle Fork Smith River, California during early August, 1993 ('93.2) and late July, 1994 ('94.2). Dashes indicate data were not available.

HABITAT	1993		1994	
	Chinook	Steelhead >1	Chinook	Steelhead >1
Pool	--	--	-0.18	*0.87
Pool Head	-0.05	0.38	-0.48	-0.49
Pool Body	0.15	*0.95	-0.15	*0.95
Pool Tail	-0.48	-0.40	-0.59	0.04
In-pool Run	--	--	0.65	-0.17
Riffle Margin	*0.68	*0.59	*0.88	*0.68
Riffle Midchannel	0.41	0.56	0.45	*0.93
Run Margin	0.26	0.13	0.04	0.30
Run Midchannel	-0.06	-0.05	0.07	0.43

* Significant correlation; $p < 0.05$, one-tailed test.

Table 7. Estimated total fish abundance (\hat{Y}) and sampling variance (\hat{V}) from both mean per unit (M.P.U.) and ratio estimation (Ratio) for selected habitat types of the Middle Fork Smith River, California with the percent change imparted by the ratio estimate.

		1993			1994		
		M. P. U.	Ratio	% Change	M. P. U.	Ratio	% Change
Steelhead >1 in Pool Bodies	\hat{Y}	2685	2050	-20	1997	2185	9
	\hat{V}	2610487	755719	-71	606841	96718	-84
Steelhead >1 in Riffle Margins	\hat{Y}	1012	1057	4	2728	2585	-5
	\hat{V}	58467	29913	-49	506517	283242	-44
Chinook in Riffle Margins	\hat{Y}	230	240	4	281	293	4
	\hat{V}	11130	8738	-21	15450	5671	-63

Estimated Total Chinook Abundance

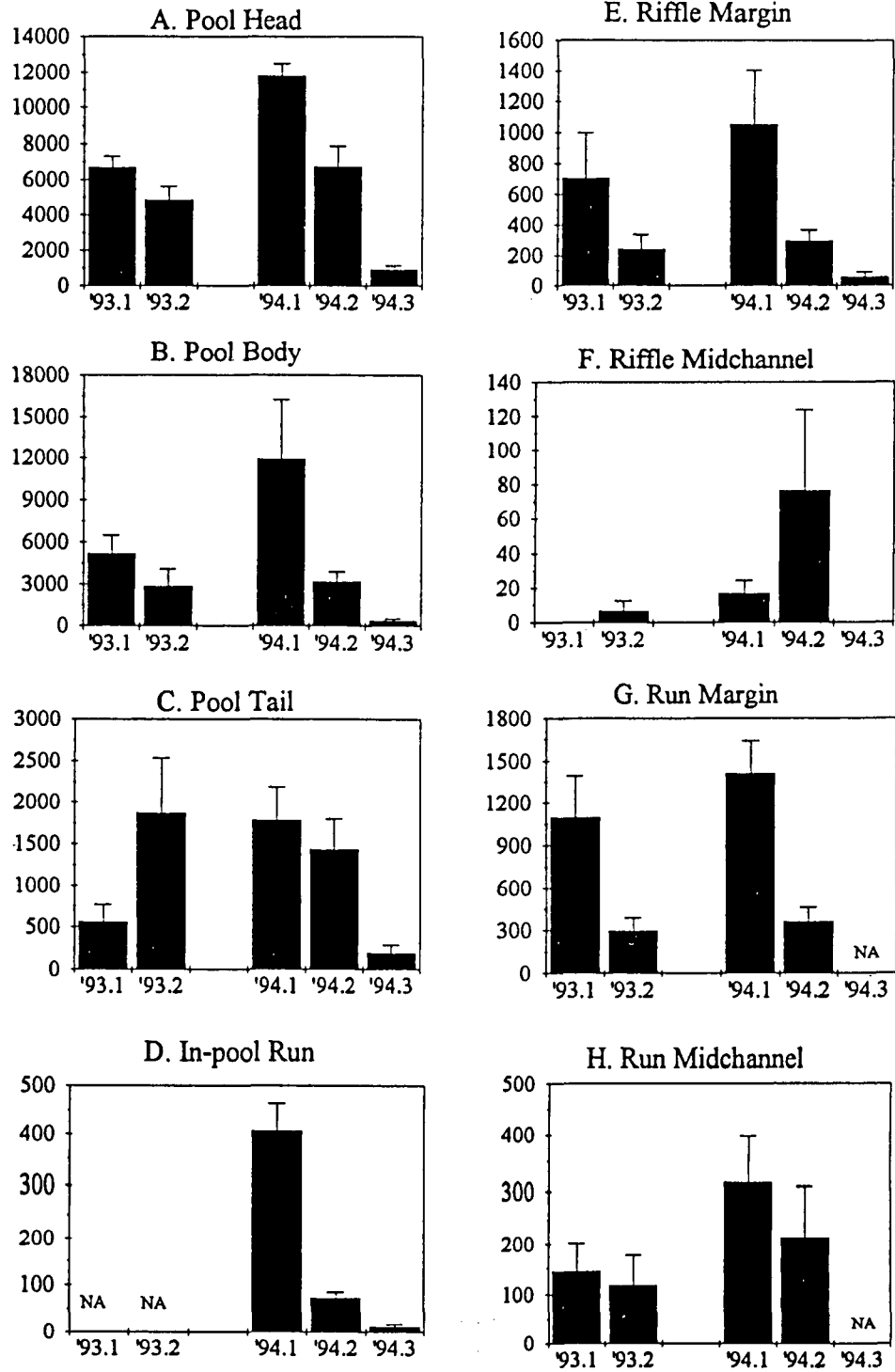


Figure 8. Estimated total juvenile chinook abundance in the Middle Fork Smith River, California with standard error bars. Graphs A - H correspond with habitat types as labeled. Five summer sampling occasions are represented in each graph.

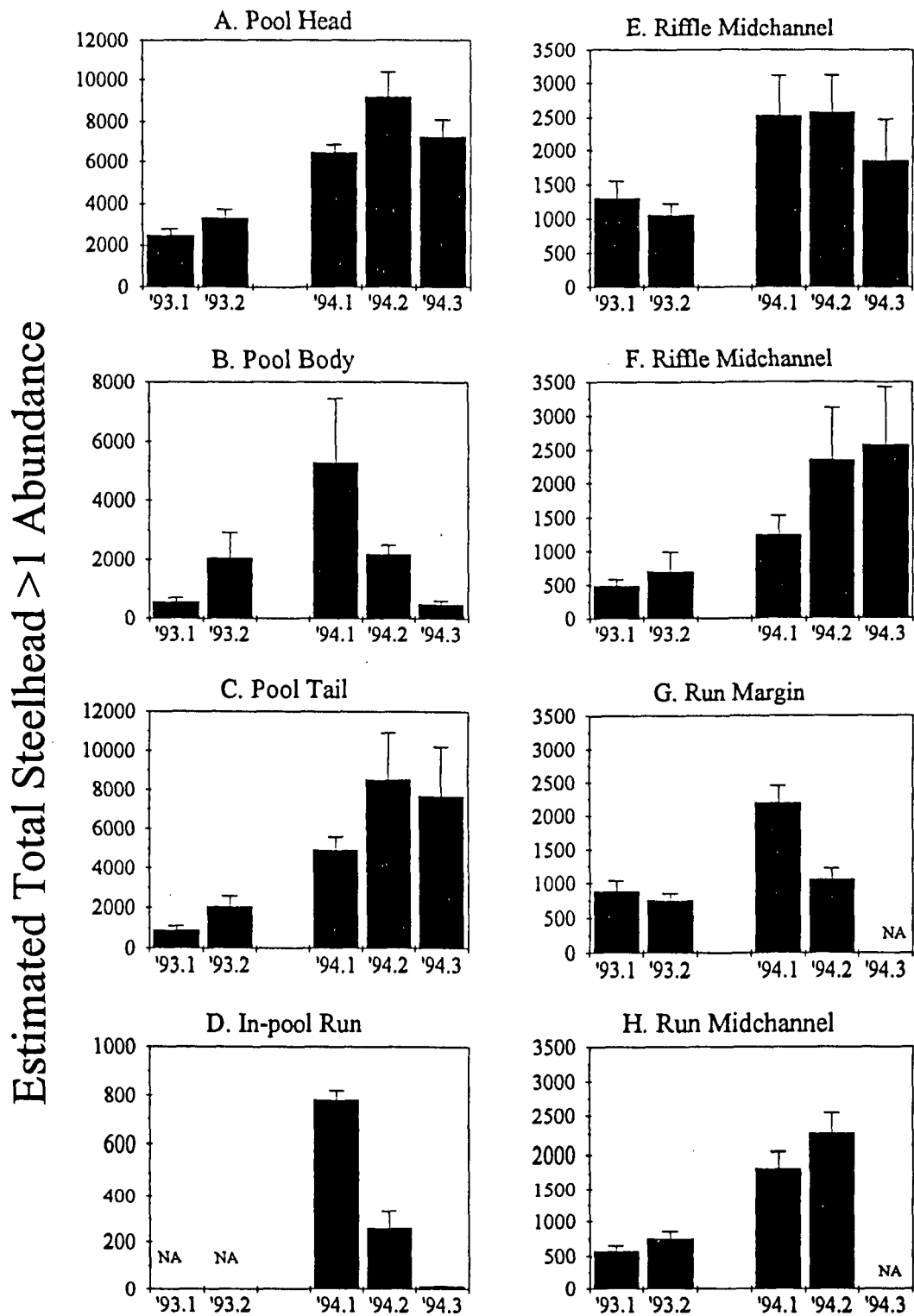


Figure 9. Estimated total steelhead >1 abundance in the Middle Fork Smith River, California with standard error bars. Graphs A-H correspond with habitat types as labeled. Five summer sampling occasions are represented in each graph.

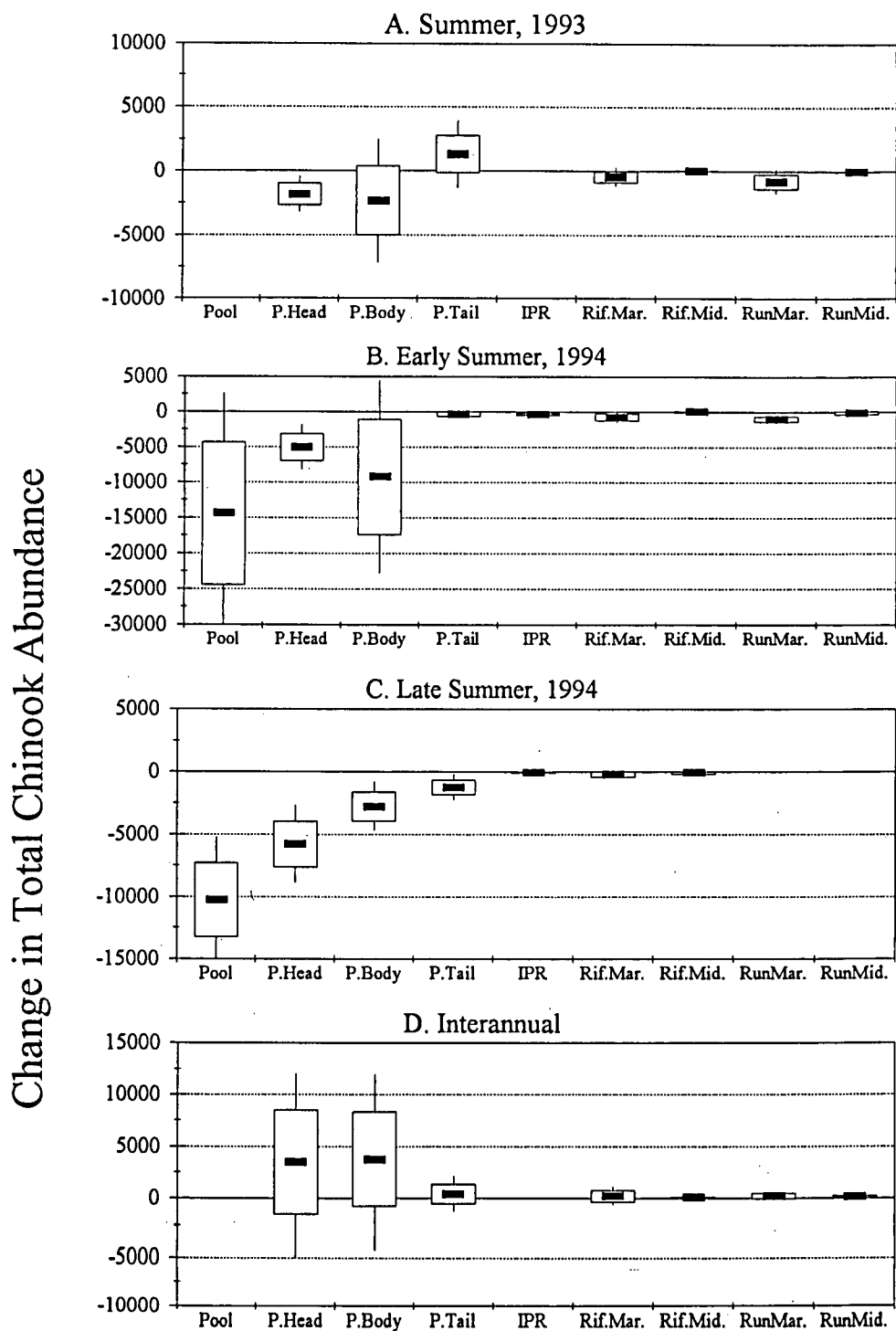


Figure 10. Estimated change in chinook abundance by habitat type in the Middle Fork Smith River, California. Estimates are bound with 90% (bar) and 98.75% (line) confidence intervals, the latter used to infer statistical significance when excluding zero. Negative values indicate decreases in abundance. Change in chinook abundance is presented in (A) as differences between '93.1 and '93.2, in (B) as differences between '94.1 and '94.2, in (C) as differences between '94.2 and '94.3, and in (D) as differences between the averages of both sampling occasions in 1993 and the first two in 1994.

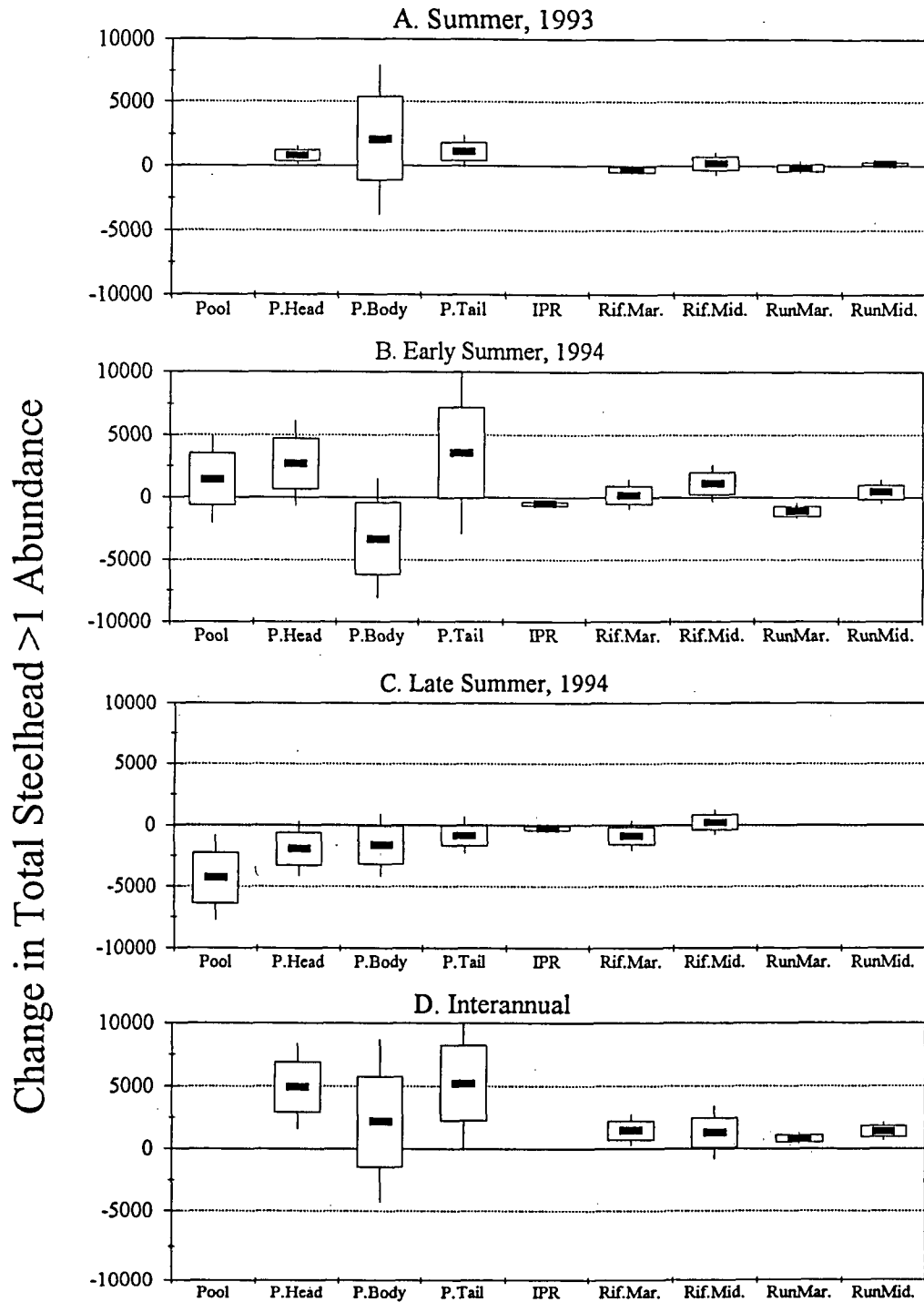


Figure 11. Estimated change in steelhead >1 abundance by habitat type in the Middle Fork Smith River, California. Estimates are bound with 90% (bar) and 98.75% (line) confidence intervals, the latter used to infer statistical significance when excluding zero. Negative values indicate decreases in abundance. Change in steelhead >1 abundance is presented in (A) as differences between '93.1 and '93.2, in (B) as differences between '94.1 and '94.2, in (C) as differences between '94.2 and '94.3, and in (D) as differences between the averages of both sampling occasions in 1993 and the first two in 1994.

Relative Observable Fish Densities

Observable fish density (no./m²) in '93.2 and '94.2 was calculated for each habitat type (Table 8). ANOVA was used to test for different observable fish densities among habitats, and groups of habitats when variances were equal ($F_{\max} < F_{.01}$) (Figure 12). All density data were log-arcsin-square root transformed before analysis. Only this composite transformation equalized variances in some cases.

Statistical comparisons of chinook densities were complicated by substantial inequality of variances. Chinook were scarce in riffle midchannel habitat (one chinook in only one sample unit during '93.2; chinook present in only two sample units during '94.2) resulting in exceptionally low sampling variance. Even with this habitat type omitted, variance of chinook density in pool tails was 14-20 times the variance for chinook density in fastwater habitats (Table 8). Variances were not unequal, however, among subsidiary pool habitats, or among fast-water habitats. Chinook densities in '93.2 differed among pool habitats (ANOVA; $P=0.0037$), and less strongly among riffle margins, run margins, and run midchannels ($P=0.0494$). Density in pool heads was significantly greater than density in pool bodies ($P=0.0107$), but neither was different from density in pool tails (two-way t-tests; $\alpha' = 0.0167$).

In '94.2, the sampling variance for chinook density in pools was approximately ten times the variances for chinook density in fast-water habitats (Table 8). A Bonferroni adjusted T-test ($\alpha' = 0.0125$) showed density in pools was significantly greater ($P=0.0101$) than density in run margins, the fast-water habitat type of highest estimated density (equal variances for this particular comparison). Chinook density

Table 8. Estimated densities (no./m²) of observable chinook and steelhead >1 in the Middle Fork Smith River, California during late summer of 1993 and 1994. Standard error in parentheses.

Habitat	Chinook Density (no./m ²)			Steelhead >1 Density (no./m ²)		
	'93.2		'94.2	'93.2		'94.2
Pool	NA	NA	0.1062 (0.0245)	NA	NA	0.1198 (0.0184)
Pool Head	0.1347 (0.0235)		0.2075* (0.1824)	0.0893 (0.0104)		0.2640* (0.1156)
Pool Body	0.0330 (0.0148)		0.0734* (0.0620)	0.0197 (0.0070)		0.0241* (0.0181)
Pool Tail	0.1364 (0.0567)		0.0931* (0.0710)	0.1493 (0.0359)		0.4716* (0.3280)
Riffle Margin	0.0120 (0.0046)		0.0123 (0.0038)	0.0678 (0.0122)		0.1825 (0.0384)
Riffle Midchan.	0.0002 (0.0002)		0.0021 (0.0016)	0.0298 (0.0076)		0.1183 (0.0256)
Run Margin	0.0208 (0.0065)		0.0241 (0.0068)	0.0430 (0.0076)		0.0949 (0.0205)
Run Midchan.	0.0099 (0.0042)		0.0156 (0.0082)	0.0670 (0.0120)		0.1781 (0.0282)

a) Estimates from a sample of seven pools which contained all three subsidiary habitat types.

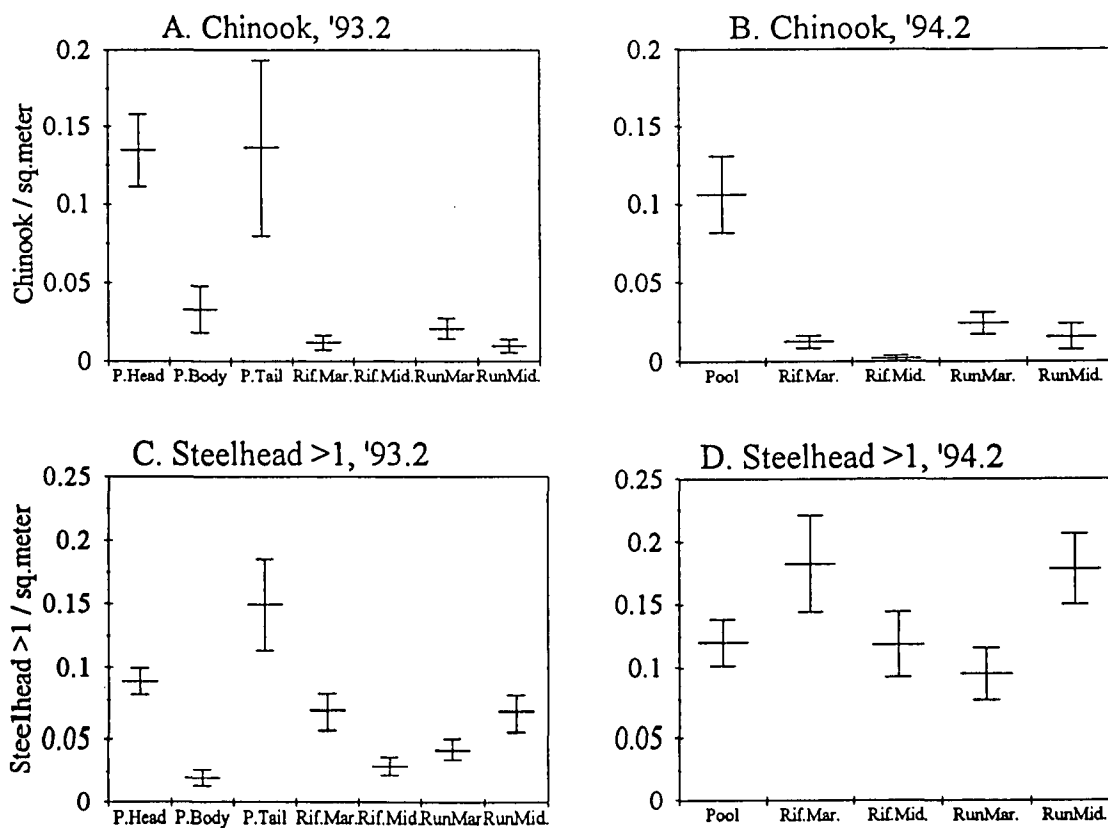


Figure 12. Mean observable fish density by habitat type with standard error bars. Densities are for juvenile chinook and steelhead >1 in the Middle Fork Smith River, California during late summer of 1993 and 1994.

did not differ significantly (ANOVA; $P=0.8718$) among riffle margins, run margins, and run midchannels.

Steelhead densities in '93.2 differed among the seven habitat types (ANOVA; $P=0.0046$). Differences existed among subsidiary pool habitats ($P=0.0070$), and less strongly among the four fast-water habitats ($P=0.0201$). By multiple t-tests, pool heads ($P=0.0018$) and pool tails ($P=0.0048$) had greater densities than pool bodies, but did not differ from one another ($P=0.1856$). Density did not differ significantly ($P=0.1309$) among all five habitat types in '94.2 (Figure 12).

Densities in subsidiary pool habitats in '94.2 were compared (excluding in-pool runs due to small sample) using paired t-tests ($\alpha' = 0.0167$). Of the ten pools sampled, nine contained head and body habitat units, and seven contained head, body, and tail habitat units. Chinook density was greater in heads than in bodies ($P=0.0143$), but could not otherwise be differentiated within pools. Steelhead >1 density was greater in heads than in bodies ($P=0.0002$), and greater in tails than in bodies ($P=0.0053$), but there was no difference in density among head and tail habitat types ($P=0.3010$).

In order to examine the relationship among subsidiary pool habitats, I calculated correlations in fish density for various pairs of subsidiary pool habitat types. Four pairs per species were tested, including density in pool heads to the combined densities of body and tail. No correlations were significant ($P>0.10$).

Paired And Repeated Counts

Paired counts were conducted predominantly on margin-type units in 1993. A fourth snorkeler necessary for paired counts in pools was seldom available. Most paired counts were made early in the summer to give snorkelers information on relative performance. Although no particular selection method was used, I made an effort to choose units that varied in form and fish abundance. Snorkelers felt that, in general, paired counts were biased by the proximity of another snorkeler which impeded view and mobility. No difference ($P > 0.4$; T-test) was detected in paired count CVs between the '93.1 sampling occasion ($n=11$) and the '93.2 sampling occasion ($n=6$). Data were thus pooled. Average CVs for paired counts were 0.109 for chinook and 0.085 for steelhead (Appendix C).

Paired visual estimates separated by 10 min. to four hours (repeats) were conducted as time allowed. Units were selected systematically (e.g., every other unit) on most days yielding an essentially random subsample per habitat stratum. Good representation of all habitat types and all levels of fish abundance was obtained within 142 total repeats (Appendix D). Pooled average CVs for all repeats were 0.198 for chinook and 0.158 for steelhead. Snorkelers generally agreed that chinook were more difficult to quantify due to schooling behavior.

Repeats for margins in '94.2 were confounded by the participation of a new snorkeler. Evident throughout that sampling effort was disagreement between the pair of snorkelers on margin boundaries. The relatively high average CV for that group

(Table 9, B) reflects error associated with boundary delineation in addition to observer variance. The '94.2 margin group was therefore omitted from subsequent analyses. Pooled average CVs for all repeats, but omitting this group, were 0.151 for chinook and 0.128 for steelhead.

Average repeat CVs for eight species-habitat groups separate into two levels of magnitude. While average CVs for most of the groups ranged from 0.08 to 0.13, the average CVs were greater (0.19 to 0.26) for chinook in pool tails and both species in pool bodies (Table 9, A, last column). There was no pattern in CV over time from the limited data. If no pattern actually existed, flow did not affect repeatability within the range encountered.

Repeats from only three other studies were available for comparison to my results. CVs for repeats in the Middle Fork were greater than those in two streams with flow less than $1 \text{ m}^3/\text{s}$, but less than CVs for repeats in the upper Sacramento River, California, where flow was of a similar range (Table 10). Data from the upper Sacramento River were for rainbow trout (*Oncorhynchus mykiss*) visually classified as longer than 10.16 cm (4") and shorter than 35.56 cm (14"). Depth sometimes exceeded maximum visibility (approx. 4 m) in the upper Sacramento River (TRPA 1994).

Table 9. Mean coefficients of variation (CV=SD/mean) for repeat counts (A) and revisit counts (B) by stratum and sampling occasion for chinook and steelhead >1 in the Middle Fork Smith River, California. Sample sizes are in parentheses. Pooled averages for all occasions are in last column.

A. Repeats						
	'93.1	'93.2	'94.1	'94.2	'94.3	All
Chinook						
Pool	NA	NA	0.0883 (5)	0.1016 (7)	0.1868 (4)	0.1118 (17)
Pool Heads	0.0469 (3)	0.0565 (8)	0.0748 (6)	0.2048 (8)	0.1965 (5)	0.1220 (30)
Pool Bodies	0.0942 (5)	0.0163 (2)	0.1245 (4)	0.3346 (7)	0.3481 (4)	0.2153 (22)
Pool Tails	0.0250 (3)	0.1009 (3)	0.3740 (4)	0.1675 (5)	0.2357 (4)	0.1923 (19)
Margins	0.0851 (6)	0.0658 (8)	0.1380 (16)	0.4918 (16)	NA	0.1087* (30)
Steelhead >1						
Pool	NA	NA	0.0128 (5)	0.0869 (7)	0.0815 (4)	0.0984 (17)
Pool Heads	0.0377 (3)	0.0376 (8)	0.1218 (6)	0.0981 (8)	0.0978 (5)	0.0806 (30)
Pool Bodies	0.1895 (5)	0.2189 (2)	0.2124 (4)	0.2441 (7)	0.4178 (4)	0.2552 (22)
Pool Tails	0.0078 (3)	0.1316 (3)	0.1104 (4)	0.1034 (5)	0.0826 (4)	0.0919 (19)
Margins	0.1593 (6)	0.0259 (8)	0.1472 (16)	0.3569 (16)	NA	0.1175* (30)
B. Revisits						
	'93.1	'93.2	'94.1	'94.2	'94.3	All
Chinook						
Pool	NA	NA	0.2617 (2)	0.3558 (3)	0.1892 (2)	0.2813 (7)
Pool Heads	0.1209 (3)	0.0677 (3)	0.0989 (2)	0.4621 (4)	0.3200 (3)	0.2381 (15)
Pool Bodies	1.0000 (1)	0.1817 (3)	0.3755 (2)	0.2930 (4)	0.6561 (3)	0.3499 (13)
Pool Tails	0.1993 (2)	0.5357 (4)	0.6653 (1)	0.2092 (3)	0.2193 (3)	0.3456 (13)
Margins	0.1131 (6)	0.3518 (6)	NA	NA	NA	0.2325 (12)
Steelhead >1						
Pool	NA	NA	0.0403 (2)	0.2457 (3)	0.0298 (2)	0.1253 (7)
Pool Heads	0.1116 (3)	0.0590 (3)	0.1231 (2)	0.4192 (4)	0.0465 (3)	0.1716 (15)
Pool Bodies	0.0667 (1)	0.0748 (3)	0.4425 (2)	0.5298 (4)	0.4786 (3)	0.3639 (13)
Pool Tails	0.0149 (2)	0.1873 (4)	0.0689 (1)	0.1104 (3)	0.0931 (3)	0.1122 (13)
Margins	0.1014 (7)	0.2086 (7)	NA	NA	NA	0.1550 (14)

a) Average excluding values from '94.2

Table 10. Coefficients of variation (CV=SD/mean) for repeat fish counts by snorkelers in various studies. Stream size is described by what was available from the following: stream order, basin area (km²), and flow (m³/s) encountered during fish surveys.

CV	Fish	Location	Stream Size	Source
0.111	coho 0+	Mill Cr., California	3rd order; < 1m ³ /s	Joe Scriven ^a
0.110	chinook 0+	Jackson Cr., Oregon	5th order; 373 km ² ;	Roper et al., 1994 ^b
0.119	steelhead >1	"	0.43 - 0.92 m ³ /s	"
0.151	chinook 0+	Middle Fork Smith, Ca.	6th order; 803 km ² ;	this study
0.128	steelhead >1	"	3.4 - 17.8 m ³ /s	"
0.119	chinook 0+	Middle Fork pools	"	"
0.098	steelhead >1	"	"	"
0.190	10-36 cm rainbow trout	upper Sacramento, Ca.	1106 km ² 1.5 - 13.5 m ³ /s ^c	TRPA 1994 ^b

a) Personal correspondence; HSU Fisheries.

b) CVs made available through personal correspondence with lead author.

c) Stream order and basin area were unavailable.

Within repeats, second counts were neither greater nor less than first counts ($P > 0.05$ for both species; two-tailed paired T-test). The lack of difference persisted within stratified habitats. There was no indication, therefore, of displacement of fish by snorkelers. Contrary to the scenario wherein snorkelers cause a fright reaction that affects subsequent counts, the means for second counts were slightly greater than the means for first counts (Appendix D).

Revisits were conducted on a randomly selected subsample of units from each habitat stratum in 1993 sampling occasions, but margins were omitted in 1994 due to time constraints (Appendix E). Mean CVs for revisits were generally greater than mean CVs for repeats and differed less among habitat types (Table 9). The pooled mean CVs for all revisits were 0.291 for chinook and 0.199 for steelhead.

Variances between repeat and revisit CVs were unequal and remained unequal after arcsin square root transformation. Variance of revisit CVs was greater for chinook and variance of repeat CVs was greater for steelhead. Revisit CVs were significantly greater, in general, than repeat CVs when a T-test for unequal variances was used ($P < 0.001$ for both species). The extent to which average revisit CVs exceeded average repeat CVs was greater for chinook than for steelhead > 1 in most habitat types (Figure 13). Fish counts in pools were generally less variable than fish counts in subsidiary pool habitats during the same occasions (Figure 13). Fish counts in pool bodies were much more variable than fish counts in pools.

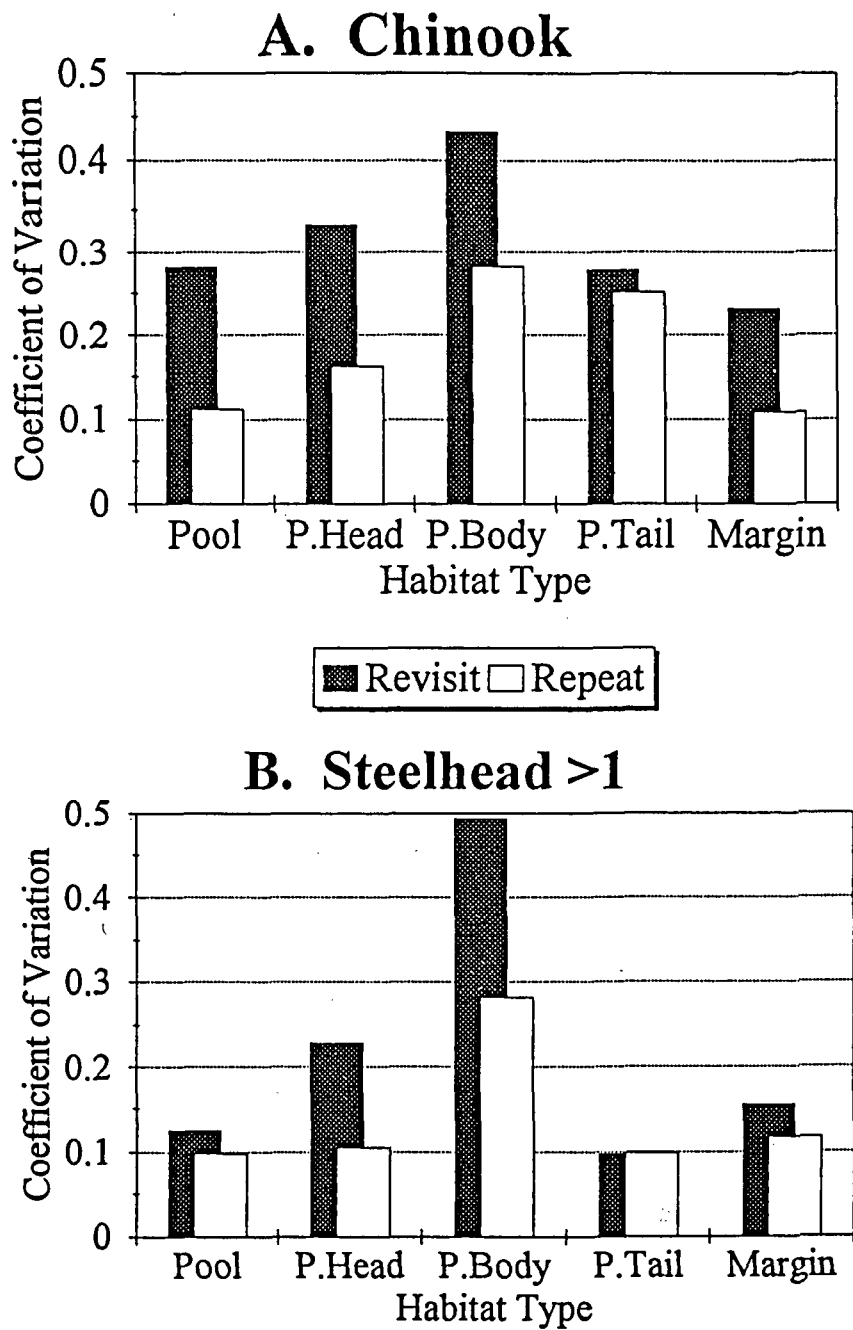


Figure 13. Mean coefficient of variation ($CV=SD/mean$) for repeat and revisit counts in the Middle Fork Smith River, California by habitat type for chinook (A) and steelhead >1 (B). For pool heads, pool bodies, and pool tails, data were pooled over all time occasions in 1994. For margins data were pooled from '93.1, '93.2, and '94.1

Greater revisit CVs indicated that changes in fish abundance per habitat unit occurred between days. To examine the effects of this sort of temporal variation on abundance estimation, I reduced my data set to those units for which both repeats and revisits were made (Appendix F), and compared two estimates of abundance for each unit. The first estimate (y_i') is the average of a repeat. The second estimate (y_i'') is the average of a repeat and a revisit count. The inclusion of the revisit count changed estimated abundance more for chinook than for steelhead (Figure 14). The inclusion of a revisit count usually increased variance of the abundance estimates, and the proportion of units for which variance increased was higher for chinook (78%) than for steelhead >1 (65%).

Repeats were used to test the hypothesis that pool tail abundance was overestimated due to the displacement of fish from pool bodies into pool tails by downstream swimming snorkelers. During the '94.2 sampling occasion, ten pool tails were subject to repeats using the standard method for one count and an alternative method for the other. Order was randomly assigned. The alternative count occurred after either no prior sampling of the associated pool body (snorkelers walked around the pool body) or after a 15-50 minute waiting period. No statistically significant difference was detected between the counts of each method (paired T-test; $P=0.40$ for chinook and $P=0.53$ for steelhead), and, thus, there was no evidence that observed pool tail densities were biased by displacing fish out of the pool body.

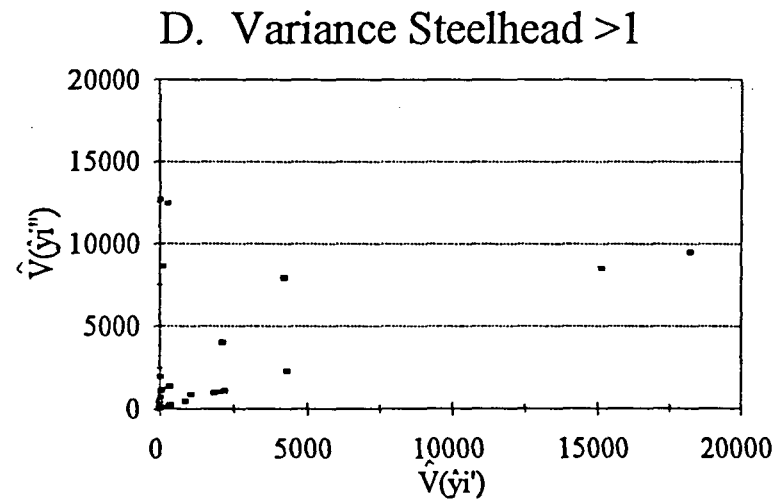
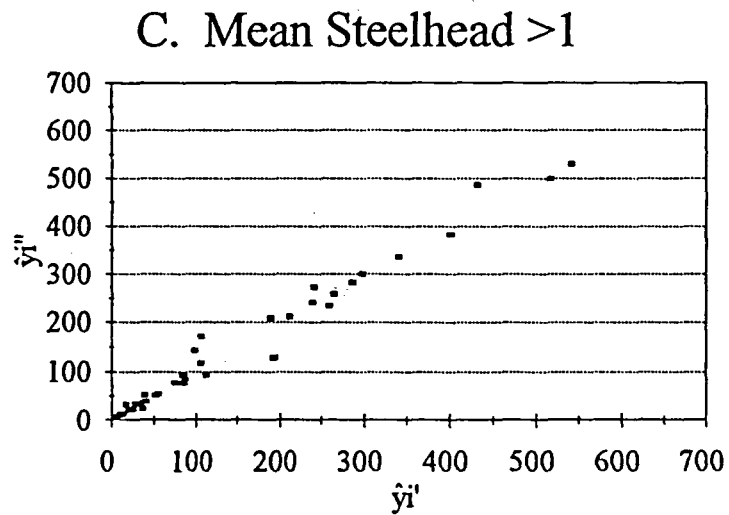
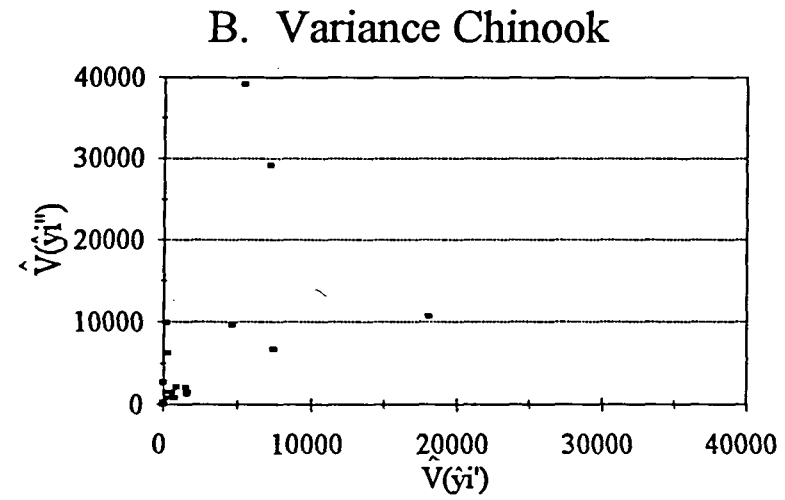
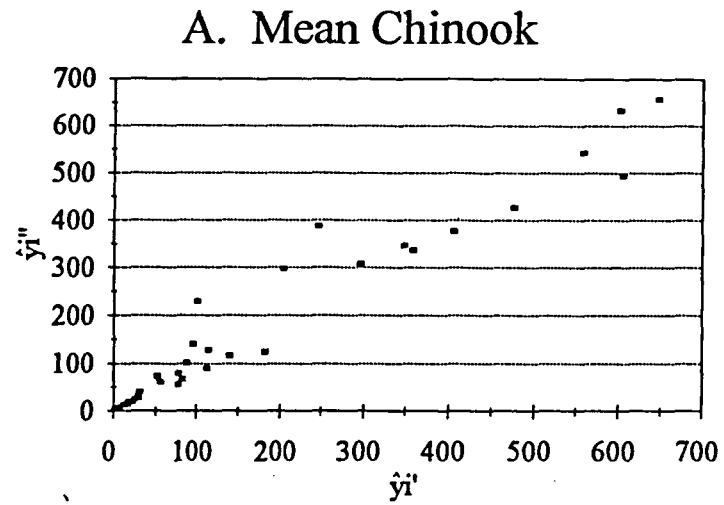


Figure 14. The relationship of two estimates of y_i and $\hat{V}(y_i)$ for chinook (A-B) and steelhead >1 (C-D) in the Middle Fork Smith River, California. The x-axis estimates (\hat{y}_i^1) are the average of two repeat counts. The y-axis estimates (\hat{y}_i^2) are the average of two repeats and a revisit (1-4 days later) count.

Repeats were also used to test whether estimates of fish abundance in pool bodies were influenced by the direction in which snorkelers swam lanes. During the '94.2 sampling occasion seven pool bodies were subject to repeats using one count by each direction and separated by a 15 minute waiting period. Snorkelers swam downstream first in three units, and upstream first in the four other units. No statistically significant difference was detected between the counts of each method (paired T-test; $P=0.32$ for chinook and $P=0.15$ for steelhead) and therefore no evidence existed suggesting one method was more bias than the other.

Emigration

Trapping occurred at Site 12 km on 50 nights during 1994. Traps were always positioned in the thalweg, but on early occasions (June 16 - June 28) were submerged entirely below the surface. Substantially lower catches of chinook on those dates probably resulted from migration of that species at the water surface (Healey, 1991). Numbers of captured chinook increased dramatically on June 29 after adding flotation to the net (Appendix G).

Estimated total emigrants past Site 12 km were plotted for 40 nights in 1994 between June 29 and August 30 (Figure 15). Fish were caught during the nighttime period preceding each date. Two fyke-nets, strapped side-by-side were operated on all but one date for which estimates were made. Separate estimates of flow and fish capture

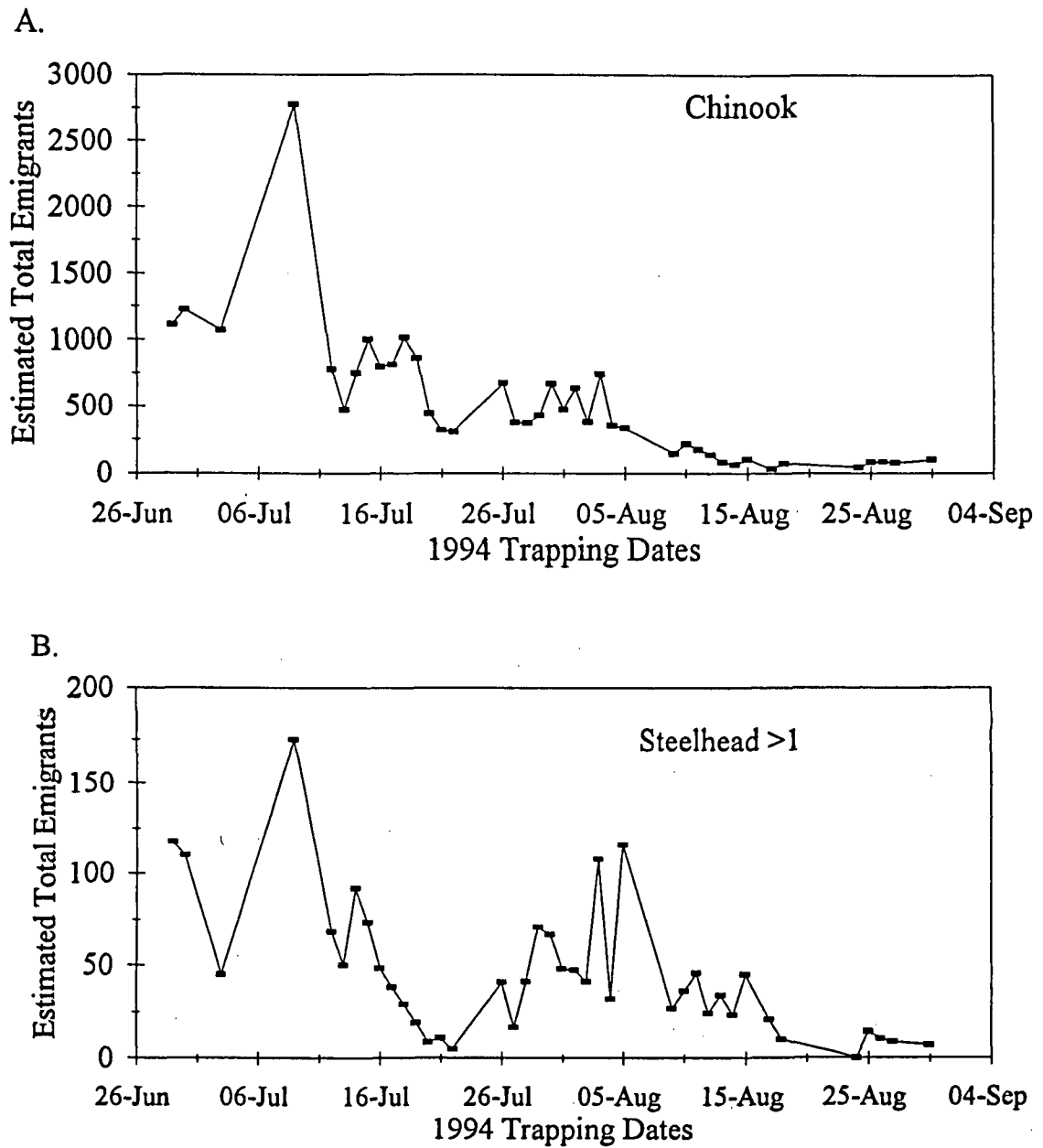


Figure 15. Estimated total chinook (A) and steelhead >1 (B) emigrants past the Site 12 km on the Middle Fork Smith River, California during nights between June 29 and August 30, 1994.

per net are available, but the difference in flow for any occasion was small, so the pair of nets was treated as one in estimating total emigrants (Appendix G).

Numbers of emigrating chinook were variable from night to night, but generally decreased from early July through August (Figure 15). Conspicuous declines in emigration rate are evident on July 20 and August 9. Levels of emigration were calculated for the periods before, between, and after July 20 and August 9 which roughly corresponded with the three fish abundance sampling occasions in 1994. Levels of emigration declined by an order of magnitude from the first to the last sampling occasion (Table 11).

Numbers of emigrating steelhead >1 varied from 0 to 175 fish per night with no clear pattern over the period of trapping (Figure 15). Emigration was low from July 20 to July 22 and again in late August. Mean numbers of emigrants for three summer periods (Table 11), were lower in late summer than in early summer. Relatively little emigration for either chinook or steelhead >1 occurred from July 20 to July 22. The moon was in full phase on July 20 and may have allowed fish to see and evade the net. Trapping notes indicated no other source of bias on these particular dates.

Table 11. Mean number of estimated total emigrants per night past the Site 12 km, Middle Fork Smith River, California for three periods in 1994. Standard deviation in parentheses.

Period	Mean Chinook per Night	Mean Steelhead >1 per Night
June 29 - July 19	1057 (552)	72 (43)
July 20 - August 9	465 (146)	47 (33)
August 9 - August 30	101 (50)	22 (14)

No fish marked and released at Site 12 km were recaptured or sighted in subsequent snorkel surveys. Of the 130 chinook marked and released at Site 0 km on July 29-31, 1994, some were found to remain at the trap site and one was recaptured at Site 12 km 2-4 days after its release. On July 31, two marked chinook were recaptured at Site 0 km. Additionally, two marked chinook were seen by snorkelers at Site 0 km on August 1, and one on August 9. These results indicated that while locally residing fish were not captured at Site 12 km, some chinook captured at Site 0 km were residing locally. The one chinook recaptured downstream migrated at a rate of approximately 4 km/day.

Emigrant Length and Growth

The frequency distributions of fork length of steelhead trapped at Site 12 km are bimodal (Figure 16). The low points within the distributions were interpreted as the minimum size of steelhead >1 . These points occurred at 75 mm, 90 mm, and 95 mm from June, July, and August, respectively, and concurred well with the criteria used by snorkelers. The length distributions indicated minimal overlap in the size of underyearling steelhead and steelhead >1 , and no separation of the multiple age classes within steelhead >1 . Assuming visual estimates of size were accurate, misclassification of steelhead >1 according to size was probably a small source of error.

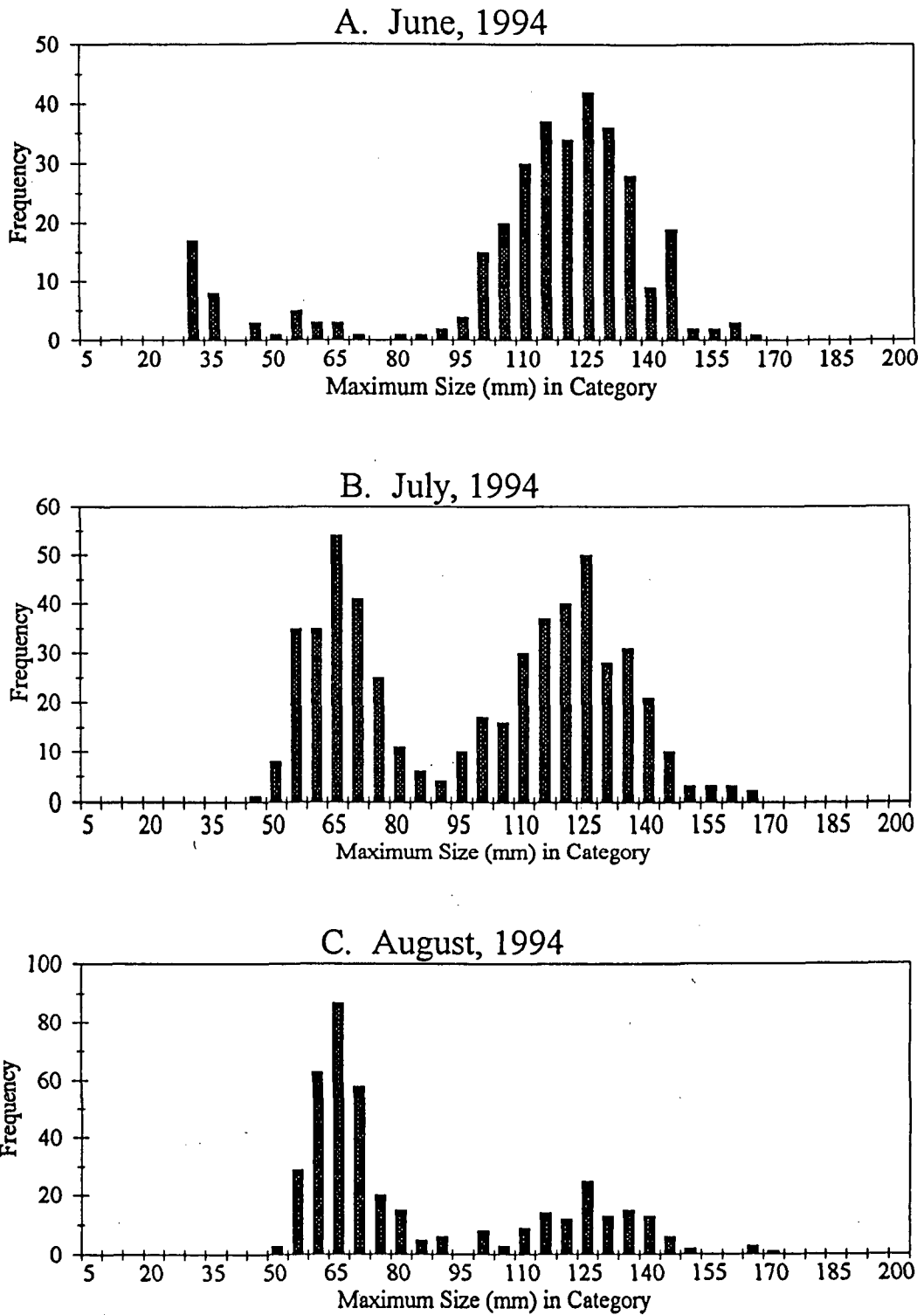


Figure 16. The lengths of steelhead captured at Site 12 km on the Middle Fork Smith River, California, during (A) June, (B) July, and (C) August of 1994.

Mean fork length of chinook and steelhead >1 trapped at the Site 12 km in 1994 was calculated and plotted for each catch of more than seven fish (Figure 17). Steelhead considered underyearling (< 80 mm in June, <90 mm in July, and <100 mm in August) were removed from the data. The mean length of steelhead >1 varied from 115 to 128 mm, and showed no pattern over the summer. Variance in length was greater for steelhead >1 than for chinook (SD ranged 10-23 Vs 1-10; Appendix H, I) due to the multiple age classes represented and small sample sizes ($8 < n < 37$). Mean length of chinook increased over summer from 61 mm in early June to 78 mm on August 30. Mean length of chinook actually peaked on June 22 at 83 mm (SD=5.2) and remained near 75 mm during July. The mean length of chinook trapped in the last week of September (not plotted) was 88 mm (SD=7.4; Appendix I).

Growth of chinook during use of the Middle Fork was interpreted from the difference between length of chinook captured at each trap site within a week. Such interpretation of steelhead lengths was deemed inappropriate due to multiple age classes represented. Chinook captured at Site 12 km were consistently larger than chinook captured at Site 0 km (Figure 18). Mean length differences were statistically significant ($P < 0.05$) for all pairs but the second week in August when only twelve chinook were captured at each site (Appendix J). Difference in length was on average, 5.1 mm and 3.3 mm in 1993 and 1994, respectively.

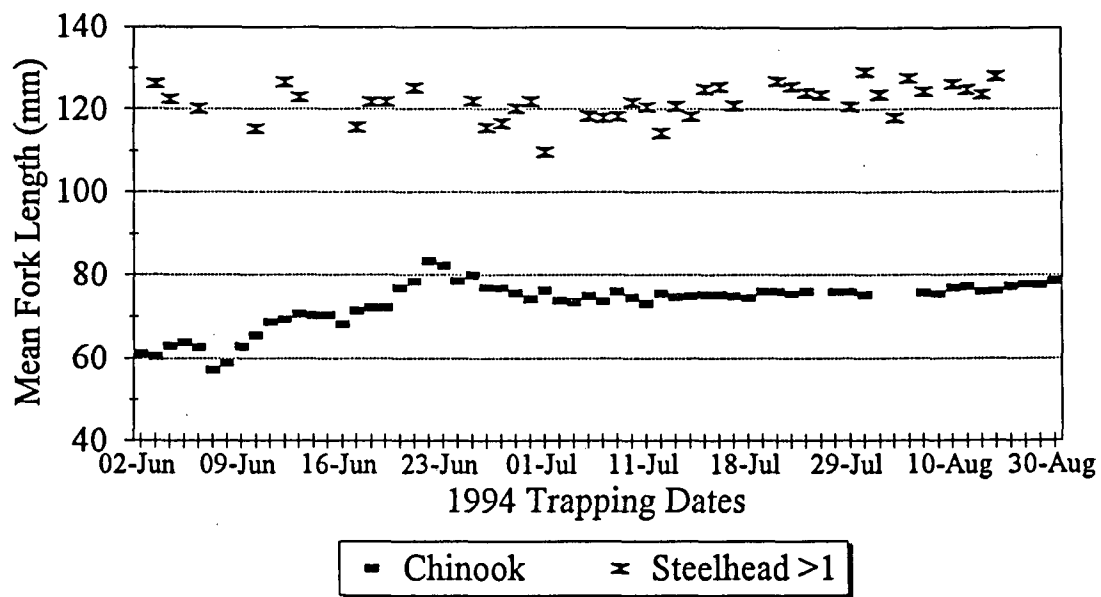


Figure 17. Mean fork length of chinook and steelhead >1 for each trapping occasion at Site 12 km, Middle Fork Smith River, California in 1994. Data from captures of less than eight fish were excluded.

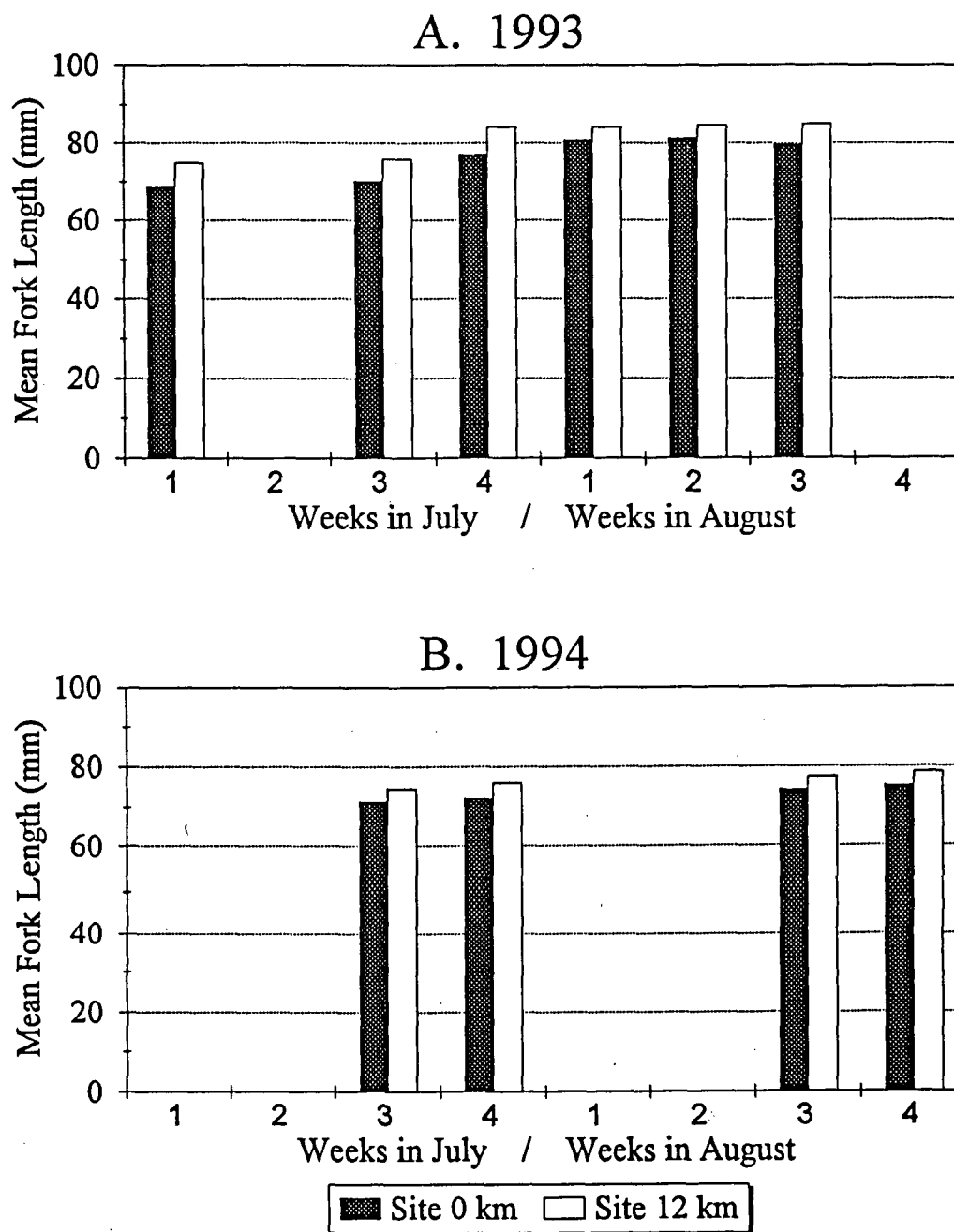


Figure 18. Mean fork length of chinook trapped at two Middle Fork Smith River, California sites during the weeks of July and August, 1993 (A) and 1994 (B). Within each week, Site 0 km was fished one to four days following Site 12 km.

DISCUSSION

Methodology

Current programs to monitor the abundance of juvenile salmonids in river basins are restricted to small streams (Dollof et al., 1993) and the published literature reveals that most of what we currently know about juvenile salmonid distribution comes from studies that have been conducted on small streams (e.g., Bisson et al. 1988, Nickelson et al. 1992, McCain 1992). A quantitative method for monitoring the abundance and studying the distribution of juvenile salmonids in mainstems is needed for assessment of basin-wide population levels, study of basin-wide patterns in distribution (Roper et al. 1994) and establishment of links between landscape level processes and variation in the life-history (Schlosser 1991). For mainstems with good water clarity, snorkeling is a practical technique for enumerating and observing fish, but methodological issues pertaining to habitat classification, measurement error, and appropriate estimation require careful consideration.

Habitat Classification

Habitat classification facilitated my objectives by providing: (1) a framework for stratified sampling of fish abundance, (2) separation of factors influencing the errors in visually estimating fish abundance, and (3) the principle resolution at which distribution (i.e., relative fish densities) was examined. In developing a general classification scheme

to serve many purposes, Hawkins et al. (1993) assert that it should be based on measurable variation in environmental attributes at spatial scales important to the activities of the stream biota. Channel units decidedly represent the appropriate scale for general habitat classification in small streams; environmental variation is conspicuous at the scale of channel units, and channel units are of relatively homogenous depth and velocity (Hawkins et al. 1993). In the Middle Fork, however, variation in depth, velocity, turbulence, and fish density was conspicuous within channel units (Figure 7, Table 8).

The hierarchical framework for stream habitat classification (Frissell et al. 1986) suggests that a variety of spatial scales can be used as units of observation and that each corresponds with a temporal scale determining rate of formation and disturbance. Below the scale of channel units, microhabitat subsystems are patches of relatively homogenous substrate, depth, and velocity that vary over days, weeks or months (Frissell et al. 1986). In studies of stream fish, the scale of microhabitat has been used most often to develop and test models of habitat availability (e.g., Bovee and Milhous 1978). Microhabitat has become synonymous with focal point data, and a scale too small for studies of abundance and distribution.

In a hierarchy of spatial scale, a reference level does not adequately represent an attribute of interest if too small, and ignores pattern in that attribute if too large (Levin 1992). The scale of microhabitat used by those who model habitat availability is inappropriate for studying distribution and abundance of fish because points in the stream

are not used by groups of interacting fish (Shirvell 1994). Channel units are the inappropriate scale of habitat classification in large streams if they repeatedly contain similar patterns of fish distribution. By this criteria, classification of head, body, and tail habitat within pools represented the appropriate scale for studying distribution of juvenile chinook and steelhead >1 in the Middle Fork because consistent, and in some cases, statistically significant differences in observable fish density were found among subsidiary habitat types (Figure 12). Additionally, the subsidiary habitat scale detected within-pool patterns in species composition, and fish behavior.

Although subsidiary habitats are at a lower level of scale than channel units, area measurements of subsidiary habitats in the Middle Fork (Table 3) are larger than most channel units in small streams. Size of channel units increases rapidly with stream order (Platts 1978), but size of fish changes very little. Lowering the scale for habitat classification in mainstems can be viewed as an adjustment to maintain the scale at the level of fish. Using the same logic, subsidiary habitat classification in small streams may not detect patterns of fish distribution within channel units because such patterns can only exist in pools of sufficient size.

Measurement Errors

The total errors of estimation in this sampling survey may be viewed as some combination of sampling variance, and measurement error. Sampling variance alone was used to estimate variance, and generate confidence bounds on abundance estimates. Some consideration of measurement error is required for assessing the utility of visual

estimates of fish abundance. Measurement errors (excluding bias) have been considered a negligible contributor to the total error of estimates of juvenile salmonids in small streams (Dollof et al. 1993), but have also been shown to be influenced by target species, temperature, visibility, and stream size (Hillman et al. 1992, Schill and Griffith 1984, Northcote and Wilkie 1963) warranting special consideration within each new application.

Bias is typically an important component of measurement error. Cochran (1977, p.377) defines bias as the difference between a true value and the expected mean of all measurements of that value:

$$B_i = Y_i - E(y_{ir})$$

where y_{ir} = r th measurement (count) in unit i .

In this study the target value or the number of observable fish in a unit, is equivalent to the expected mean of all counts in that unit. Thus, bias cancels to zero and measurement error is simply the difference between a count and the expected mean of all counts:

$$e_{ir} = y_{ir} - E(y_{ir})$$

where e_{ir} = error for unit i on the r th repeat occasion.

The above equation demonstrates that the errors for a particular unit have the same frequency distribution as repeat counts for that unit. Variance of repeat counts estimates measurement error not associated with bias or when bias is zero. Although

repeats were a sample of only two, the mean CV of repeats from many habitat units provides an index of measurement error for the group those units were sampled from.

Relatively high mean CVs (0.22 for chinook; 0.26 for steelhead >1) for repeats in pool bodies (Table 9, A) indicated relatively high measurement error for this habitat type. The deepest water of some pool bodies was often poorly lighted. The effective range of visibility in those locations was sometimes less than depth. Pool body was the only habitat type wherein variability of repeat chinook counts was less than variability of repeat steelhead >1 counts. Steelhead >1 frequently were observed to use deep water. Chinook were more strongly associated with the water surface.

Mean CV for repeat chinook counts in pool tails was relatively high (0.19), but mean CV for steelhead >1 in pool tails was low (0.09)(Table 9, A). Density of chinook in pool tails was low relative to density of steelhead >1 and mixed schools were commonly encountered. Under these conditions, snorkelers commonly overlooked chinook as underyearling steelhead, or incorrectly identified them as steelhead >1.

Mean CVs for repeat chinook counts in pool heads were high in 1994 (>0.19), but not in 1993 (<0.08)(Table 9, A). Higher measurement error in 1994 may have resulted from higher densities of chinook in pool heads during that year (Table 8). By contrast, mean CVs for repeat steelhead >1 counts in pool head and pool tails were low (< 0.10), despite being associated with the highest of estimated fish densities (Table 8). I attribute these results to differences in fish behavior. Chinook in pool heads were best characterized as dynamic aggregations or swarms that often measured 3-4 m in height,

while steelhead >1 usually maintained position relative to one another and near (< 1 m) the bed. Schooling, and territoriality appeared, thus, to be aspects of fish behavior which affected measurement error.

In summary of CVs for repeats among species-habitat groups (Table 9, A), measurement error varied among habitats and between species, but in each case of conspicuously high measurement error, fish behavior was implicated along with some other factor. Depth, misidentification of fish, and high fish density all appeared to increase the errors of visually estimating fish abundance in units of certain habitat types. Within each of these habitat types, however, there were different affects on measurement error between species, and these differences can be explained by behavioral characteristics. Pool bodies were the only habitats wherein visual estimates of fish abundance were, in general, excessively variable, and this was likely a function of depth.

Mean CVs for repeats in margins varied among sampling occasions (Table 9, A). The CVs for repeats in '94.2 (0.49, 0.36) showed that measurement error can be very high when snorkelers disagree on margin/midchannel boundaries. On other sampling occasions mean CVs for repeats in margins were low (< 0.09), indicating that the subjective boundaries between margins and midchannels may not be a large source measurement error if observers perceive boundaries similarly.

Analysis of the difference between repeat observations indicated that fish displacement was not a large source of measurement error in this study. Fish were infrequently observed fleeing from snorkelers. While displaced groups of fish were

sometimes observed in pool bodies, such displacement was not seen in pool heads or margins where feeding by fish seemed to predominate regardless of snorkeling activity. A monitoring of fish displaced by snorkelers in the upper Sacramento River (TRPA 1994) revealed that downstream displacement out of a unit was a problem when the next unit downstream was deep, but virtually nonexistent when the downstream unit was predominantly whitewater. I believe that, in this study, such a phenomenon contributed error to counts in pool tails that were followed by deep riffles, but that such error could be minimized by low-impact procedures, or the use of blocking nets.

Measurement error, in this study, can be partitioned into the following sources: movements of fish in and out of cover, movements of fish in and out of habitat units, observer performance (e.g., missed fish and double-counting fish, misclassification of fish), displacement, and ambiguous boundaries. The last two sources appear to only operate for certain habitat types or habitat units. All but the first two sources can be reduced through snorkeler training, and carefully selected, consistent procedures.

Movements of fish in and out of cover was not specifically addressed in this study, but was implicated by effect of depth in pool bodies. Depth functioned as cover in some pool bodies, and some variation in the use of deep water by fish certainly occurred. Variation in the use of other types of cover (e.g., substrate, debris, turbulence) would also be a source of measurement error. Visual estimates of rainbow trout in the upper Sacramento River, California were greater during afternoon than during morning (TRPA

1993). Diel variation in repeat counts was not measured in this study, but probably occurred as a function of the proportion of habitats illuminated among other factors.

Movement of fish in and out of habitat units was likely to occur due to boundaries of classified habitats that did not discourage fish movements. Movements between subsidiary pool habitats were indicated by lower mean CVs for repeats in pools in 1994 (0.12 for chinook; 0.10 for steelhead >1) than mean CVs for repeats of the subsidiary habitats of those pools (Figure 13). The indistinct boundaries of pool bodies did not restrict fish movement as much as the boundaries of pools. Movement of fish in and out of margins was not specifically assessed, but CVs for margin repeats were not particularly high, indicating that such movement was not a large source of measurement error.

One noteworthy aspect of observer variance was variability in size classing fish. As summer progressed, underyearling steelhead became more difficult to differentiate from steelhead >1 (Figure 16). I believe that the separation of size classes was not consistent among snorkelers during '93.2, when a new and relatively inexperienced snorkeler was used to count fish in pools. Both lack of experience classifying fish by size underwater, and numerous size classes likely increase the role of variable size classification in measurement error. Variable size classification was one of the largest sources of measurement error when others visually estimated abundance of four size categories of rainbow trout in the upper Sacramento River (TRPA 1994).

The contribution of measurement error to the total errors of estimation can not be determined without a more accurate method of estimating fish abundance. Dollof et al. (1993) used the relationship between estimates from multiple pass electrofishing and paired visual estimates to suggest that, in small streams, the contribution to the total errors of estimation by variability in visual estimates is negligible relative to the contribution by sampling variance. With this assessment as a benchmark, comparison of mean CVs of repeats provided a means of evaluating how significantly measurement error compromised bounds of confidence on abundance estimates in this study.

Mean CVs of repeats in this study were not much greater than those measured in smaller streams, and less than those measured in the Upper Sacramento River (Table 10). The upper Sacramento River does not provide a good comparison because snorkelers in that study were counting resident fish of four different size classes and, visibility varied spatially and temporally. The difference between mean CVs in this study and those from smaller streams may be accounted for by movement of fish across the quasi-boundaries unique to this study. Mean CVs for repeats in pools during 1994 were not higher than mean CVs for repeats in smaller streams, indicating the variability of snorkeler counts within similarly defined habitat units was not greater in the Middle Fork Smith than in smaller streams.

Monitoring Fish Abundance

The utility of fish abundance estimates lies, not only in the reliability of measurements, but, moreover, in the ability to detect actual changes in abundance over

time. Given a certain region of rejection (e.g., $\alpha=0.05$), the ability to detect an actual difference between two abundance estimates, or the power of that comparison, is determined by sample size and the precision of the estimates (Dowdy and Wearden 1985). For uncalibrated visual estimates of fish abundance in the Middle Fork, the ability to detect actual change should also be evaluated with regard to the assumptions of constant proportional bias and a fixed object of estimation.

Sampling Precision

Sampling precision (inverse of sampling variance) is a function of sample size and variance. Increases in sampling effort will always lead to increases in sampling precision for finite populations (Cochran 1977). Sampling precision may also be increased by methods of estimation that utilize a relationship between the target variable and an auxiliary variable if a sufficiently strong relationship exists. Probability proportional to size (PPS) and ratio sampling are two such methods. Although more efficient than ratio estimation (Hankin 1984), PPS restricts analysis of distribution, and prohibits sampling frame adjustments due to unequal probability selection.

In the Middle Fork, sufficiently strong correlation between chinook abundance and habitat area were found only in riffle margins, and sufficiently strong correlation between steelhead >1 abundance and habitat area were found only in riffle margins and pool bodies (Table 6). The ratio estimator generated lower variance estimates in cases (Table 7), but these gains should be qualified by the bias of the estimator. Cochran (1977) states ratio-based estimates of sample variance are biased by the order of $1/n$ and

generally unsatisfactory for $n < 12$. The average percent bias for eight natural populations was -21% for $n=8$ and -18% for $n=12$ (Rao 1969). In the Middle Fork, all but one ratio-based estimate deviated from the unbiased mean per unit estimate less than 5%. For steelhead >1 abundance in pool bodies during 1993, the ratio-based estimate deviated by -20%. Only seven pool bodies were sampled in 1993, whereas sample size equaled or exceeded ten in the other cases. In conclusion, ratio estimation increased precision of abundance estimates by approximately 50% (Table 7) without any apparent losses in accuracy when sample size equaled or exceeded ten, but bias was apparent in the case of a smaller sample size.

A strong positive correlation was found between pool area and steelhead >1 abundance (Table 6), but stratification of subsidiary habitats revealed that this was almost entirely due to the correlation between the pool body area and the number of steelhead >1 therein. The most precise method of estimating total steelhead >1 in pools would stratify the subsidiary habitats of pools and sample independently from them, assuming that the proportion of bias was constant among habitats. This would allow use of the ratio estimator for the pool body stratum and the mean per unit for the pool head and pool tail. Such a method is undesirable, however, because numbers of fish per pool would not be available and comparisons between occasions would be greatly complicated by the ratio-based estimates of variance (Raj 1968).

The strength of correlation between fish abundance and habitat area is a function of the variability in habitat area. The exceptionally variable area of pool bodies (Table 3;

CV=1.2) indicated that steelhead >1 do not necessarily associate more strongly with area in these habitats than in other habitat types. Significant abundance to area correlation in riffle margins (Table 6) may not have occurred if riffle length had been less variable (Table 3; CV=0.68). In riffle margins, and run margins, abundance correlated more strongly with length than area for steelhead >1 in both years and chinook in 1993. Fish abundance in margin habitats may be more closely associated with length than area because food delivery occurs along a lateral boundary and not across the width of the habitat unit (Moore and Gregory 1988). As an alternative to using the ratio of fish abundance to habitat area, precision in the estimation of fish abundance in margin habitats might be improved by either using the ratio of fish abundance to habitat length, or by partitioning long riffles into units of a more standard length in order to decrease the variance of fish abundance per unit.

Assuming Constant Proportional Bias

The proportion of juvenile chinook and steelhead >1 in the Middle Fork which could not be seen by snorkelers was probably small due to outstanding water clarity and the predominant feeding activity of these fish. Life history requirements (Healey 1991) and optimal temperatures for growth (Brett et al. 1982) were strong stimuli for chinook and steelhead in the Middle Fork to feed constantly during daylight hours. Fish behavior, water clarity, and light conditions in summer are factors indicating that bias was small. More importantly, however, the consistency of these factors supported the assumption of

constant proportional bias within individual habitat units over and between summer, an assumption that is fundamental to the utility of abundance estimates.

The assumption of constant proportional bias within strata is also fundamental to the utility of abundance estimates, and was supported by a habitat classification system based on channel units, depth, and velocity. Each of these physical attributes is associated with cover and fish behavior (Bisson et al. 1988, Hillman et al. 1992). The stratification of habitat by these attributes thereby provided strata that varied less (within strata) in the proportion of unobservable fish. For strata composed entirely of habitat units wherein there is an opportunity of observing every fish, the assumption was supported. For strata with the deepest and fastest water (i.e., pool bodies, midchannels), the assumption was not supported. Beyond some range, depth and velocity will influence bias, and their interaction with small and previously unimportant variation in fish behavior, and cover may have a compounding effect. With increasing velocity chinook, for example, move deeper into pools (Steward and Bjornn 1987).

Assuming a Fixed Object of Estimation

Visual estimates of fish abundance in habitat units of the Middle Fork were more variable over a period of one to four days than within a day (Figure 13). Part of this difference was attributed to seaward migrations, 95% of which may occur at night (USFWS 1995). Overnight variation in the fish abundance per unit may be interpreted as an additional source of measurement error when sampling occasions exceed one day. A

more serious implication of this temporal variation, however, is the invalidation of abundance estimates from sampling occasions that exceed one day.

Sampling theory includes an assumption that the object of estimation is fixed over the period required for sampling, and that a sampling frame consists of fixed values per sampling unit (Raj 1968). If values vary slowly, a sample survey may be conducted rapidly enough to warrant ignoring this issue. Numbers of chinook per unit exhibited greater temporal variation than numbers of steelhead >1 (Figure 13) as a function of higher migration rates (Figure 15). Estimates of chinook abundance per unit were substantially affected by the inclusion of a count made one to four days later (Figure 14). It is not clear to what extent seaward migration invalidated estimates of fish abundance, but future sample surveys of chinook abundance in the Middle Fork should be completed in as few days as possible.

Abundance

Life history requirements provide the context wherein we find meaning for changes in abundance and interpret patterns of use. A river segment and season of interest (e.g. mainstem during summer) are found at a unique place within the spatio-temporal pattern of use by each species in that river basin. This discussion will apply knowledge of chinook and steelhead life history to observations in the Smith River, so that its populations may begin to be characterized. The limited scope of this project combined with the diversity in the life history patterns of juvenile salmonids makes the

discussion somewhat speculative. I hope, however, that patterns implicated here will be tested in future projects that expand our view in space and time.

Life History and Seasonal Variation in Abundance

Migration is recognized as an important source of temporal variation for all salmonid populations (Hall and Knight 1981), and is certainly the predominant factor in the seasonal abundance of chinook and steelhead >1 in the Middle Fork. Alternatively, mortality rates for juvenile chinook are typically very high (Healey 1991). For Atlantic salmon (*Salmo salar*), predation on migrating smolts has been measured at 35% over an 8 km river segment and 50% over a 22 km river segment (Larsson 1985). Coastal cutthroat trout (*Oncorhynchus clarki clarki*), river otter (*Lutra canadensis*), common merganser (*Mergus merganser*), and great blue heron (*Ardea herodias*), were all observed to feed on juvenile salmonids in the Middle Fork.

Chinook

From Nicholas and Hankin (1988), who reviewed data available on Oregon coastal chinook, a probable life history sketch is available for Smith River chinook salmon. The vast majority of juvenile chinook enter the ocean in their first year, spending only three to six months rearing in the river. Ocean-entry typically occurs in late summer or fall after a period of estuary use that may vary from a week to five months. By early summer, chinook have dispersed throughout the mainstem network and many have reached the estuary. Healey (1991) suggests that the volitional downstream migration of 50 - 120 mm "fingerlings" in early the summer functions primarily to disperse fish into

available habitat downstream. The pattern of downstream movement during early summer is described by Nicholas and Hankin (1988) as an essentially constant "flow", but over spring and summer, the downstream movement of juvenile chinook might be described better as a "wave". On the Trinity River, California, the magnitude of seaward migration by chinook increases through March and April, peaks in May or June and decreases through July and August (Moffet and Smith 1950, USFWS 1995). Visual estimates of juvenile chinook abundance were used at established sites extending 55 miles below Lewiston dam on the Trinity River (USFWS 1986-91). At upriver sites, highest densities are seen in March or early April, but downstream densities often rise through April and into May. Densities at upriver sites decrease by an order of magnitude from spring to summer.

Juvenile chinook salmon abundance in the Middle Fork Smith River decreased over the summer months (Figure 10, A-C) corresponding with the descending limb of a "wave" of migration. Nicholas and Hankin (1988) classified the duration of riverine rearing for Oregon coastal rivers as short when essentially all migration to the estuary occurs by midsummer, and extended when juveniles are relatively abundant in mainstem segments throughout summer. The duration of riverine rearing by Smith River chinook would be classified as moderate because density in pools exceeded 0.1 fish/m² in mid-summer, but was observed to be extremely low in August 1994 and during preliminary observations in September 1992. Nicholas and Hankin (1988) believe the extent to which juvenile chinook remain in river during the summer is related to water

temperature; chinook with extended rearing patterns were found in relatively cool rivers. The relationship of water temperature and duration of riverine rearing may also exist within a river basin over time. Middle Fork juvenile chinook abundance decreased 23% over five weeks of early summer in 1994 when temperature ranged from 14-19°C, and 56% over four weeks of the same period in 1994 when temperature ranged from 16-22°C. Alternatively, the faster depopulation in 1994 could have been influenced by lower flow or higher density.

Steelhead

Juvenile steelhead in Northern California rivers emerge in the spring months and rear in freshwater one to five years before emigrating to the ocean, usually in spring (Barnhart 1986). Ocean survivorship increases with smolt size, and smolts less than 140 mm have little chance of surviving to maturity (Wagner et al. 1963, Ward and Slaney 1988). Due to this size requirement, length of freshwater residence is a function of growth rate (Peven et al. 1994). Generally, the larger parr of each year class smolt in spring and the smaller parr continue to reside in the river. Individual growth rates are influenced by position within dominance hierarchies (Abbot and Dill 1988), as well as environmental factors.

Mainstems are important for steelhead populations that have access to them. Approximately 97% of all emigrant steelhead >1 from Calf Creek (Harkleroad and La Marr 1992), and 95% from Steamboat Creek (Dambacher 1991), both tributaries of the North Umpqua River, Oregon, showed no smolt characteristics. The authors believe

these parr and many emigrating young-of-the-year continue residence in the mainstem, a strategy termed partial rearing. Partial rearing has been demonstrated as a dominant strategy in Gobar Creek, Washington (Lieder et al. 1986). In Gobar Creek, 77% of trapped emigrants were parr, most of which did not pass through a 14 km mainstem within the same year.

Migration of steelhead >1 parr into mainstem river segments during spring may be a response to the recession of flow and consequent depletion of suitable habitat area in tributaries. Dambacher (1991) documented the concentration of steelhead >1 into mainstem segments of Steamboat Creek, Oregon as flows receded, but the role of flow was confounded by lower temperatures in Steamboat Creek than in some of its tributaries. In the Middle Fork, moderate increases in steelhead >1 abundance were seen during summer of 1993, but not in early summer of 1994 (Figure 11, A-B) when flow had recessed more completely by the beginning of summer (Figure 6). Steelhead >1 abundance decreased moderately during late summer 1994 (Figure 11, C). The small number of steelhead >1 emigrants during this period (Figure 15, B) indicates that migration into the Middle Fork was very low during late summer when flow had long since dropped to low levels (Figure 6).

Steelhead abundance in the Middle Fork remained high throughout each of two summers. Neither significant immigration nor emigration from the study segment was indicated by comparisons between periodic abundance estimates (Figure 11) or trapping (Figure 15). These results, compared to patterns elucidated in other streams (Dambacher

1991, Harkleroad and La Marr 1992), suggest many of the steelhead observed in the Middle Fork immigrated there from tributaries during flow recession in the spring, and resided there throughout summer. Low numbers of steelhead >1 captured on fall trapping dates (Appendix G), and a dominant life history of seaward migration in the spring (Barnhart 1986, Shapovalov and Taft 1954) indicate that some steelhead >1 may have resided in the Middle Fork throughout winter.

Annual Variation

Year to year variability in juvenile salmonid abundance can be extremely high due to variation in the number of spawning females and the role of density-independent factors which cause high mortality at the egg and alevin life stages (MacKenzie and Moring 1988). Variability in annual abundance decreases with the age of the cohort due to the regulating influence of density-dependent factors such as food and space (Chapman 1966). Chinook fry abundance varies by two orders of magnitude, and chinook fingerling abundance by one order of magnitude (Healy 1991). Index estimates of annual abundance aim to monitor this variation in annual parr abundance, and could be useful in studying the relative roles of escapement, bed scour, and density dependent growth and survival in the production of juvenile chinook and steelhead.

Annual differences in abundance of juvenile chinook were confounded by seasonal variability. My approach in averaging two sampling periods from each year sought to include some seasonal variability in order to minimize the chances of a type I error. As a consequence, interannual changes in chinook abundance were not statistically

significant. Steelhead abundance, in contrast, showed significant change in many habitat strata. Comparison of the first sampling periods of each year would have resulted in more statistically significant differences for chinook. Within the context of seasonal variability, however, single occasions offer weak inference regarding annual abundance levels.

Growth and Emigration

The assumption that fyke-nets did not capture locally residing fish was supported for Site 12 km, but not for Site 0 km. Although trapping at both sites occurred in the thalweg of pool heads, the depth of water differed substantially between sites. Site 12 km was shallow (1-2 m) in all points of the pool head on all trapping occasions. Site 0 km, in contrast, was a deep (3-4 m), laterally scoured pool with a strong eddy that conveyed water from downstream to upstream of the fyke-net. Bias from capturing locally residing fish at Site 0 km limited the inference from differences between size of captured fish at each site.

Chinook

In coastal rivers of Oregon during summer, juvenile chinook differ in size between upstream and downstream sites (Nicholas and Hankin 1988). The size difference is a function of growth during downstream migration. An additional factor may be faster migration by larger fish. In the Middle Fork, the size difference of captured chinook was greater in July than in August of each year (Appendix I). This

pattern may have resulted from faster migration rates (Healey 1991) or greater metabolic demands in response to increased water temperature (Crowder and Magnuson 1983). Nicholas and Hankin (1988) speculated that growth rate of chinook in mainstems is influenced by density. This may be supported by the fact that differences in size were greater in each week of 1993 than in corresponding weeks of 1994 (Appendix J). The largest chinook (100-105 mm) seen in the Middle Fork were captured in July 1993 (Appendix I). Without direct information on migration rates, however, the relative influences of temperature and density on growth could not be discerned from these data.

Nicholas and Hankin (1988) noted that any mainstem sample of chinook may include both temporary residents and migrating transients (migrants). This was the case with the sample of 110 chinook captured and marked at the Site 0 km. Two marked chinook were recaptured at the same site in subsequent days and one migrated downstream at no less than 3.5 km/day. Due to variable rearing strategies and levels of smoltification, migration rates of individual chinook in the Middle Fork probably ranged from near zero to greater than 12 km per day, the mean migration rate for naturally reared Klamath River chinook (USFWS 1995). The slow rate was evident from the chinook that remained at the trap site for nine days.

The distinction between temporary residents and migrants is artificial because some fish certainly alternate or blend the two behaviors. The distinction is useful, however, because relative degree of migration and residency influences growth rate, observed abundance, and distribution. Changes in the proportion of chinook in "migrant"

status may be represented by changes in the ratio of estimated total emigrants to estimated total chinook in the study segment for each sampling period. This ratio increased from 3.8 percent in '94.1 to 4.8 percent in '94.2 to 6.7 percent in '94.3. Interpretation was confounded, however, by temporally variable migration rates. Rogue River, Oregon chinook increased their migration rate with size and the progression of summer (Cramer and Lichatowich 1978).

Steelhead

The natural variability in size of steelhead >1 (Figure 16) was the result of multiple age classes and growth depensation or different growth rates among individuals (Abbot and Dill 1989). Small samples from this variable population precluded any strong inferences regarding growth, but I believe growth rate declined to very low levels in late summer. The increase of 4-5 °C in average daily water temperature over summer increased the metabolic cost of maintaining a constant level of feeding activity (Crowder and Magnuson 1983). The apparent response of steelhead >1 to increased temperature and decreased food delivery, commensurate with a 50% flow reduction, was to decrease feeding activity. Most steelhead observed in June held positions in or near rapidly moving water, but a large proportion of steelhead observed in August were found in dense groups either near the bed below the turbulent water of pool heads, or in pool tails.

Despite possible decreasing growth opportunities, steelhead emigration from the Middle Fork was low upon the commencement of trapping and remained low, if not lower, through summer. Trapping occasions in late September and mid-October

indicated no greater immigration or emigration during early fall. In Idaho, the movement of steelhead into mainstem segments at the onset of winter has been attributed to the requirement for large substrate for winter habitat (Bjornn 1971, Chapman and Bjornn 1978). Larger streams have larger substrate (Platts 1978), and where steelhead migrate into mainstems during spring for habitat area, they might also remain in mainstems throughout winter for the larger substrate.

Distribution

Comparisons of observed fish densities between habitat types or species are not entirely reliable in studying actual distribution due to variable proportional bias. This discussion is restricted, therefore, to comparisons wherein differences in proportional bias are believed to be small relative to differences in density. These cases were abundant due to the observable nature of juvenile chinook and steelhead >1 in the Middle Fork during summer.

Models for habitat selection by drift feeding salmonids explain some of the observed segregation of species within habitats, and relative densities among different habitat types. According to such models, territorial behavior procures the food opportunity of some space (Chapman 1966) which is selected for maximum potential net energy gain (Fausch 1984, Fausch and White 1986, Hughes and Dill 1990). In the simplest of models, net energy gain is the energy from drift food items obtained minus the energy required for maintaining position. Some additional aspects of the models

include flexibility of the minimum space requirement according to the density of food (Chapman and Bjornn 1969), and a size to velocity relationship which functions to partition habitat among different sized fish that otherwise use the same resources (Lister and Genoe 1970, Everest and Chapman 1972).

Segregation Within Habitats

Segregation between species was particularly evident in pool heads and margins, habitats largely characterized by velocity gradients. The ability of steelhead >1 to occupy faster water within any of these habitats was due their larger size and different body morphology. Steelhead are particularly suited to utilize fast water due to their cylindrical body shape and short median fins, while juvenile salmon are more suited for maneuvering due to their laterally compressed bodies and long fins (Bisson et al. 1988).

Within pool heads, most chinook were found in the deep, downstream portion of the unit where velocity was lowest. Most steelhead >1 were located close to fast water, positioned near the bed, behind boulder or bedrock protrusions, or in eddies. In margins, steelhead >1 were abundant in the margin/midchannel transition area, while chinook were most commonly found inshore. This species stratification matched a hydraulic one, and did not occur neatly in margins with large boulders or bedrock protrusions.

Relative Densities Among Habitats

Chinook density was much higher in pools than in any other habitat (Table 8).

Although margins were also characterized by slow water, chinook density in margins was an order of magnitude lower than chinook density in pools. Margin use by chinook may be relatively low due to the shallow depths, association with the bank, or lateral exchange of water that additionally characterize margins. The latter may reduce delivery of food or colonization by downstream migrants. Shallow, bank-associated water is likely to have a higher risk of predation for small fish (Schlosser 1991). Steelhead >1 density was also high in pools, and more strongly concentrated in the head and tail ends where the velocity was highest. A preference for the head portion of pools has been described for chinook (Steward and Bjornn 1987) and steelhead >1 (Dambacher 1991). Riffles are believed to produce food in excess of what can be captured by the fish rearing there, and it is, therefore, advantageous for a fish to maintain the most upstream position relative to suitable space (Chapman and Bjornn 1969).

The density of steelhead >1 in pool tails was extremely high relative to density in other habitat types in the Middle Fork (Table 8). Use of pool tails by steelhead >1 is particularly interesting when considering the downstream position of these habitats relative to other fish in the pool, and the lack of dominance hierarchies or territorial behavior exhibited by the steelhead >1 found there. Temperature has been shown to influence steelhead distribution elsewhere (Dambacher 1991), and may also play a role in pool tail use. Pool tail abundance increased over early summer in both years concomitant

with temperature, and a higher proportion of steelhead >1 in pools were found in pool tails in '94.2 than in '93.2 (Table 5) when temperatures were lower.

Pool tails may contain optimal foraging sites for steelhead > 1 in pools during summer. Because steelhead are not efficient in slow water (Bisson et al. 1988), and temperature, above some threshold, dramatically increases the cost of obtaining food (Crowder and Magnuson 1983), a positive net energy gain may be impossible for steelhead >1 in pool bodies during late summer. In pool heads, the potential for net energy gain may also be less than in pool tails due to the extra costs associated with turbulent water. The metabolic costs of turning and acceleration can be substantial component in the energy budget of fish. Krohn and Boisclair (1994) found that the turning and acceleration of spontaneous swimming brook trout cost them approximately five times the energy of fish swimming at sustained equivalent velocities. As a result of maintaining position in fast water with minimal turning and acceleration, fish in pool tails may maintain positive energy budgets on much less food than their counterparts in turbulent or tranquil habitat.

Steelhead in pool tails maintained positions close (5-20 cm) to the streambed and among each other, occasionally rising to meet drifting food items. The dominance hierarchies and territorial behavior considered important in the distribution and production of juvenile salmonids (Fausch 1984, Hartman 1965) were not observed, and the space occupied by each fish was much lower than that considered a required minimum (Allen 1969). The adoption of this unusual non-territorial feeding strategy

could be triggered by temperature, or high density. Such an alternative foraging strategy may be similar to that of the Japanese ayu (*Plecoglossus altivelis*), which is territorial for an intermediate range of density, and schools at high density (Kawanabe 1969). Mean individual growth of the ayu is actually higher at high densities due to the alternative foraging strategy. Like the ayu, suspension of territoriality by steelhead may increase mean fitness under high densities.

Habitat Shifts

Some changes in fish abundance for particular strata were counter to the trend for that species, indicating a potential change in habitat selection. Within both summers, chinook abundance actually increased in riffle midchannels (Figure 10). Chinook have been shown to move into deeper and faster water as they increase in size (Lister and Genoe 1970), and decreased water velocity as a function of flow may have accelerated this habitat shift. The abundance of steelhead >1 in run margins decreased significantly through summer. A deteriorating velocity gradient between run margins and run midchannels probably provided progressively poorer foraging opportunity.

During mid-summer, approximately 30% of riffles and 50% of runs, by area, were classified as margin habitat. Use of margin habitat is related to fish size (Lister and Genoe 1970) and complexity of the stream edge (Moore and Gregory 1988). Fish density in margins and the proportional use of margin habitat may be greater in spring than that observed in summer due to smaller fish size and less availability of low velocity water. While kayaking the Middle Fork during winter and spring, I observed that while

the proportional area of margins decreased with increasing flow, the absolute amount of margin habitat did not noticeably change. Some margins became more distinct during high flow; velocity within margins remained low, while velocity in the expanded midchannel increased. At summer low flow, margins blended with midchannels, and midchannels provided more feeding stations with suitable velocity. These stream habitat dynamics, and the possible relationship of margin utilization to fish size suggest that the importance of margins, both as habitat and sampling stratum, may peak in the spring and diminish thereafter.

Effects of Migration

Within a migration corridor, each habitat unit is potentially subject to daily changes in use as downstream migrants leave and recolonize overnight. Habitat selection by parr is influenced by complex factors including intra- and inter-specific interactions (Shirvell 1994), making consistent levels of use under a migratory flux unlikely. Redistribution of a migrating fish species occurs, therefore, each night. A population in which a large proportion is undergoing migration ought to have a more variable (temporally) distribution than a population that does not have a large proportion of migrants. This prediction was supported by the relative affects of revisit counts on estimates of chinook and steelhead abundance in individual habitat units (Figure 14), and by the following analysis of the consistency with which each species distributed from occasion to occasion.

Assuming no change in habitat attributes, the correlation of fish counts between two paired samples should be strong if fish are selecting habitat according to consistent criteria and weak if habitat selection is influenced by stochastic factors. These correlations were calculated by habitat type for each pair of sampling periods adjacent in time (one pair in 1993 and two in 1994) and indicated different habitat selection over time and between species. Strong positive correlations ($r > 0.50$) were found for steelhead >1 in most habitats at all times, but such was the case for chinook only in the late summer of 1994 (Appendix K). In late summer of 1994, the correlations for steelhead >1 were higher than for chinook in every habitat except pool bodies (Appendix K). I suggest that this species difference indicates relatively high temporal variation in the distribution of chinook, and that the higher variability is due, in part, to high emigration rates, as evidenced by a more stable distribution in late summer of 1994 when the number of emigrants had decreased (Figure 15, A).

Chinook were observed to be more gregarious than steelhead. Schirvell (1994) suggests that intra-specific interactions commonly confound efforts to link habitat selection with physical attributes. A high proportion of migrants, and a schooling rather than territorial behavior, probably contributed to the more variable distribution found for chinook.

CONCLUSIONS AND RECOMMENDATIONS

Visual estimates of juvenile chinook and steelhead >1 abundance in pools and margin habitats of the Middle Fork Smith River during summer were no less reliable than visual estimates of juvenile salmonids in small streams when experienced teams of three snorkelers were used. Variability in visual estimates of abundance were high in the deepest pool bodies (5-10 m), but this did not seriously undermine the reliability of visual estimates in pools because pool bodies had relatively low densities of juvenile fish. The reliability of visual estimates in the Middle Fork was probably due to exceptional water clarity (7-10 m) and the conspicuous nature of target fish in summer.

Snorkel surveys of fish abundance in the Middle Fork could provide a cost-efficient method of monitoring annual production of juvenile chinook and steelhead >1 in the Middle Fork basin. Compared to abundance estimates from one or two tributary streams, Middle Fork abundance estimates could more accurately reflect basin-wide trends by (1) averaging the variation among tributaries, and (2) representing the cumulative effect of unknown production levels from all habitats upstream. Due to good access, and high densities of juvenile fish during summer, surveys in the Middle Fork are likely to monitor the largest proportion of fish in the basin for the lowest cost.

Indices of annual chinook and steelhead >1 abundance may be generated from less sampling effort than conducted here. Sampling need not occur in all habitat types. Stratification of subsidiary pool habitats is unnecessary, but within each sampled pool

may contribute valuable ecological information. The proportion of fish in pool tails, or pool bodies, for example, may correlate with flow, temperature, or density.

Different characteristics of each species warrant separate approaches to monitoring annual abundance levels. Chinook abundance decreased rapidly over summer and numbers of fish per unit varied nightly due to migration. These factors represent two forms of temporal variation, one simply in the object of estimation and the other in the relationship between the sampling frame and the object of estimation. A reliable index of abundance should account for the first form of temporal variation (decreasing abundance) and minimize the effect of the second form of temporal variation (variable abundance per unit). I suggest repeat sampling for an index of chinook abundance in May, June, July, and August. Each estimate should be based on a randomly selected sample completed in one day. To generate the most reliable index in such a short time, I suggest sampling from pools only. Approximately 85% of observed chinook were seen in pools on all sampling occasions.

Steelhead >1 abundance in the Middle Fork during summer did not vary substantially, but some habitat shifts occurred. The most reliable index of steelhead >1 abundance would be based on samples from pools and runs. Sampling should occur late in summer when runs can be examined in their entirety by an upstream progressing team of snorkelers. Precision in the estimates of steelhead >1 abundance by strata would benefit from increased sampling effort, and since steelhead >1 migration is low in summer, sampling occasions could extend over many days.

Emigration should be monitored during future fish surveys in the Middle Fork to continue developing the relationship between seasonal variation in abundance, growth, and emigration. Emigration, temperature, and flow data could be used in conjunction with monthly chinook abundance estimates to develop an understanding of the factors that influence observed abundances and sizes of chinook. This information could be used to reduce the sampling occasions or sampling effort and obtain equally reliable indices of annual chinook abundance. Migration of juvenile steelhead should be studied at a basin-wide scale; emigration from tributaries and the Middle Fork should be monitored spring, summer, and fall to determine spatio-temporal patterns of rearing and smoltification. Such trapping data would provide meaning for abundance estimates beyond the monitoring of interannual variation.

Observed steelhead >1 density in the Lower Middle Fork is high compared eight streams reviewed by Dambacher (1991). High densities of steelhead >1 may be linked to the availability of winter habitat, which comes in the form of large cobble substrate and functions to provide stable interstices (Bjornn 1971, Bustard and Narver 1985). A conspicuous attribute of the Middle Fork Smith River is the size and unembedded quality of bed particles. Siltation and stream aggradation are current regional trends (Knopp MS), the cumulative effects of which are directed toward mainstem river segments. Steelhead habitat requirements, regional trends, and the threatened status of steelhead within the Klamath province (NOAA 1994) warrant more assertive watershed restoration and long-term monitoring of populations and streambed conditions in the Smith River.

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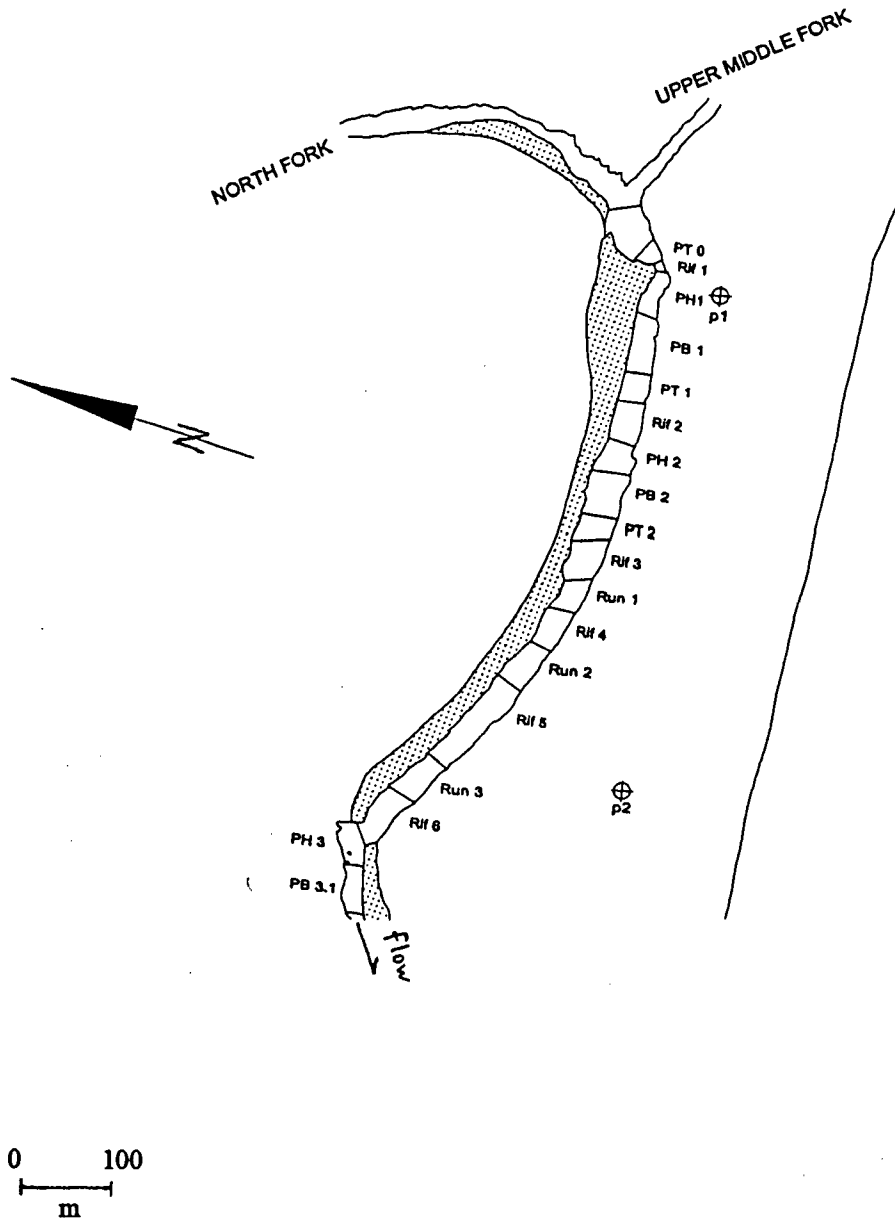
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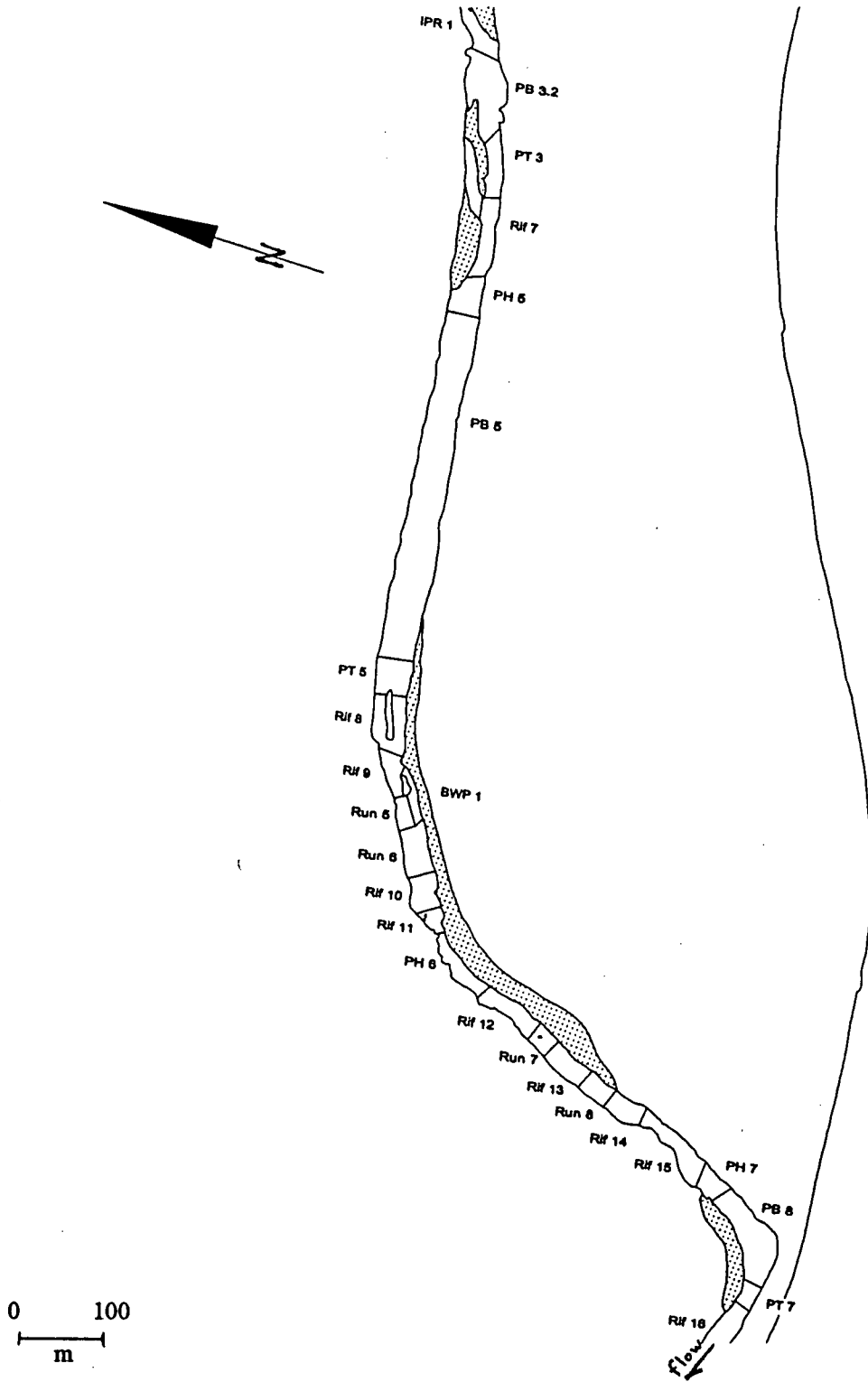
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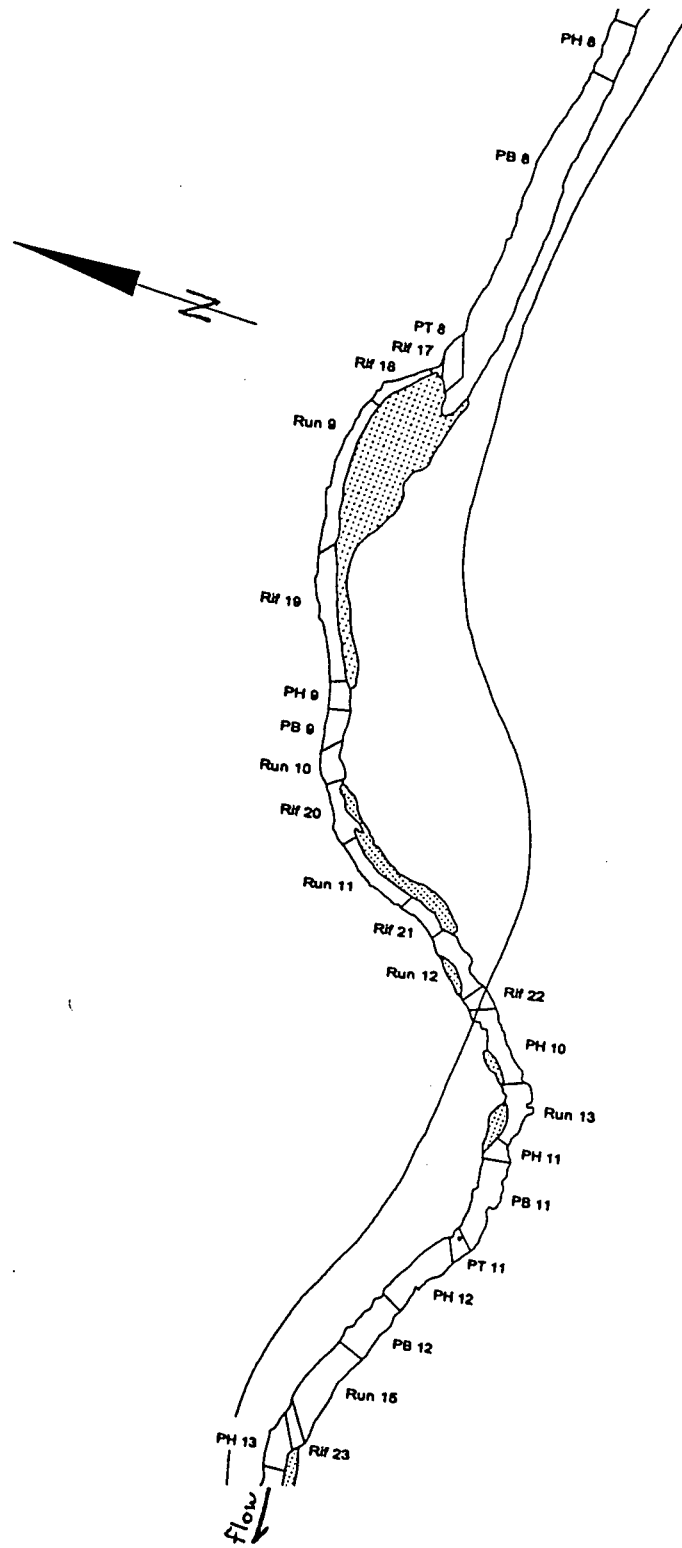
APPENDICES



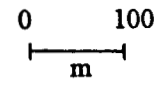
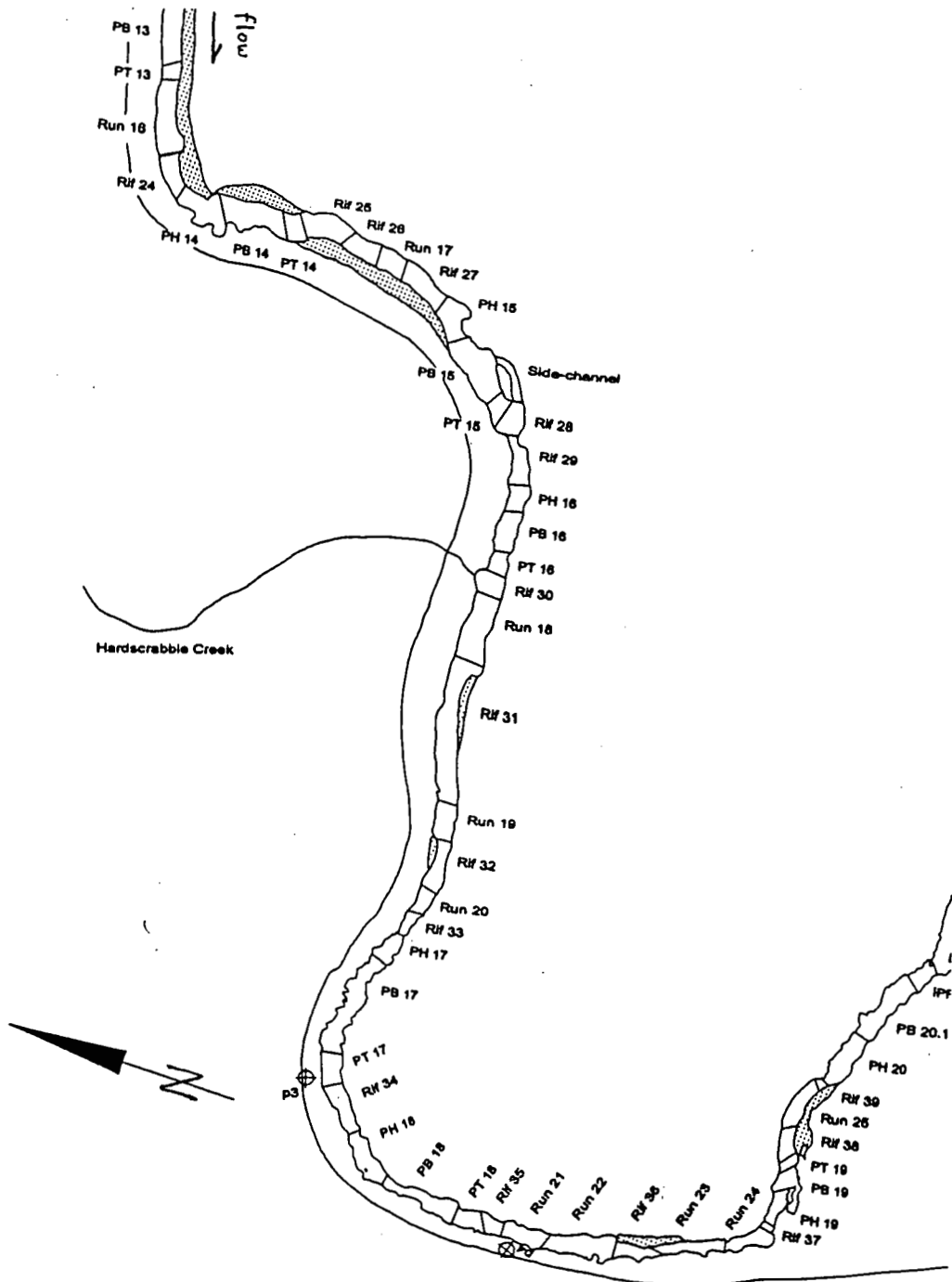
Appendix A. The Middle Fork Smith River, California from aerial photographs taken on August 18, 1994. All habitats except margins were delineated using landmarks noted in the field and hydraulic characteristics evident from the photos. Habitat labels were preserved from the original sampling frame and are abbreviated as follows: (PH) pool head, (PB) pool body, (PT) pool tail, and (Rif) riffle. Discontinuous enumeration and maximum label values exceeding N_k are due to the following sampling frame adjustments: (1) Pool 4 merged with Pool 3 by changing Run 4 and PH 4 to IPR 1, (2) Pool 23 merged with Pool 22 by changing PT 22, Rif 42, and PH 23 to IPR 4, and (3) Run 14 became part of PT 11. Six Global Positioning points used in calculating scale are represented with a crossed circle (Continued on next four pages).



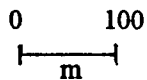
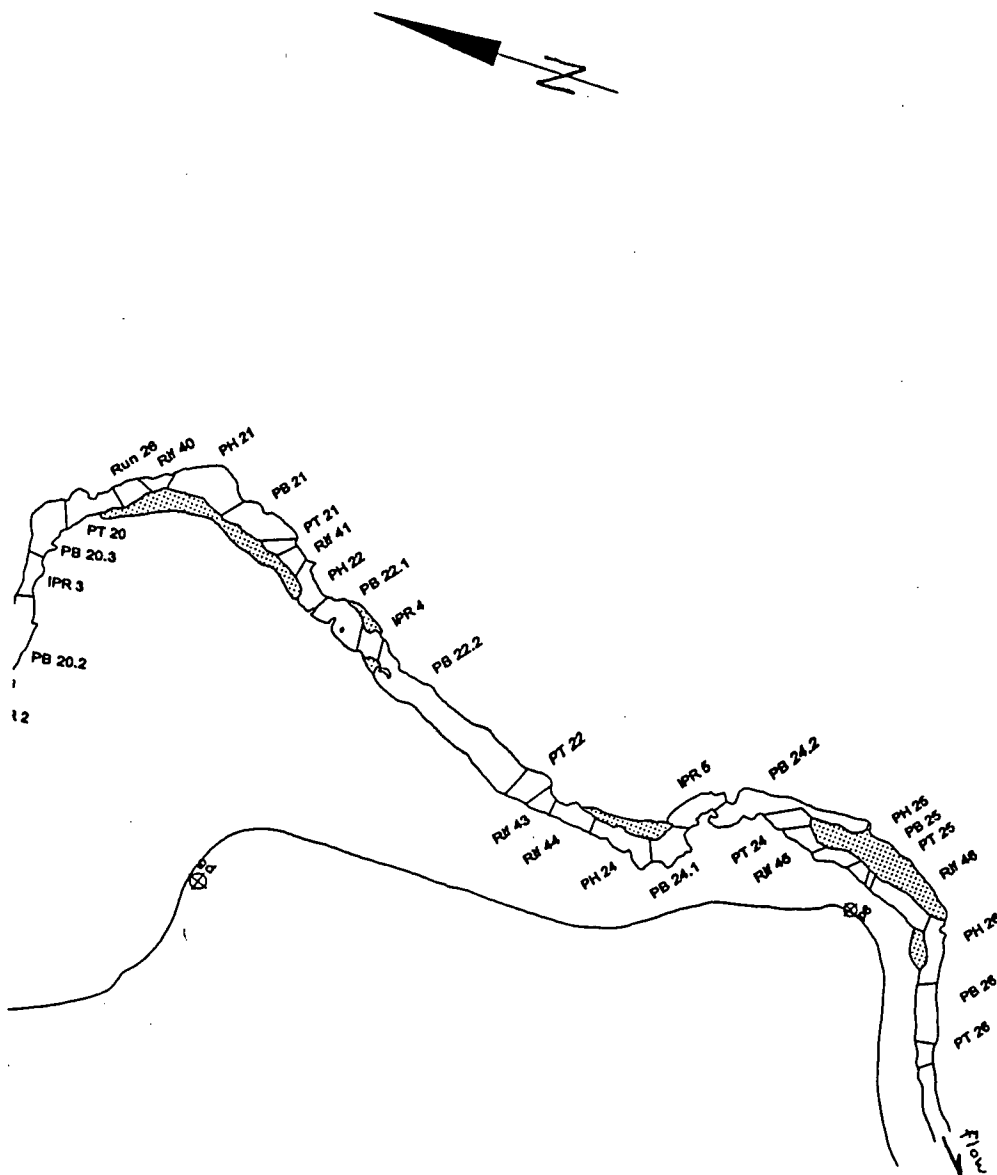
Appendix A (Continued).



Appendix A(Continued).



Appendix A(Continued).



Appendix A(Continued).

Appendix B. Counts of juvenile chinook and steelhead >1 in the Middle Fork Smith River, California by habitat unit and sampling period. Averaging of repeat counts leads to some fractions. Habitat types are abbreviated as follows: PH, pool head; PT, pool tail; IPR, in-pool run; RiMa, riffle margin; RiMi, riffle midchannel; RuMa, run margin; RuMi, run midchannel (Continued on following page).

Habitat Unit	Chinook					Steelhead >1					
	'93.1	'93.2	'94.1	'94.2	'94.3	'93.1	'93.2	'94.1	'94.2	'94.3	
Pool	2	--	--	335.5	103	11.5	--	--	309	307	233.5
Pool	8	--	--	3310	93	4	--	--	2084	2273	1742
Pool	10	--	--	540.5	315	13.5	--	--	294	242	340.5
Pool	12	--	--	712.67	168.2	1	--	--	295.33	144.8	1
Pool	14	--	--	824.5	667.33	128	--	--	664.5	516	368.33
Pool	16	--	--	778	676.5	100.33	--	--	633.5	581	448
Pool	18	--	--	1099	1026.6	138.33	--	--	651.5	1023.6	602
Pool	20	--	--	1451	856	52	--	--	1131	1108	931
Pool	22	--	--	628.5	451.5	27	--	--	405	792	501
Pool	24	--	--	1215	544	146	--	--	821	900	933
PH	2	134	89	293	49.5	8	26	36	196.5	190.5	158.5
PH	6	150	57.5	--	--	--	59.5	84.5	--	--	--
PH	8	234	85	660.5	30.5	1	203	236	342	592.5	490
PH	10	230	220	540.5	315	13.5	60	115	294	242	335.5
PH	12	186	59	348.67	121.6	1	33	20	175	122.6	1
PH	14	274	276	449.5	297.33	116.67	184	285	345.5	274	241.33
PH	16	414	131	603	501	80	106	136	307.5	402.5	336.67
PH	18	280	346	632	566.6	73	87	84	241	460	212
PH	20	512	472	458	373	39	114	223	344	581	404
PH	22	454	455	475	446.5	27	96	165	222.5	739	494
PH	24	301	126	446	101	27	100	144	233	222	339
PH	26	180	108	--	--	--	178	135	--	--	--
PB	3.1	62	20	--	--	--	18	58	--	--	--
PB	8	458	128	2622.5	47	0	3	508	1284	471	10
PB	13	149	25	--	--	--	26	20	--	--	--
PB	17	218	409	--	--	--	39	88	--	--	--
PB	20.2	294	118	226	150	0	21	9	67	23	0
PB	22.2	86	17	--	--	--	4	6	--	--	--
PB	26	78	21	--	--	--	39	7	--	--	--
PB	12	--	--	364	46.6	0	--	--	120.33	22.2	0
PB	14	--	--	225	229.33	5.6667	--	--	62.5	33	35
PB	16	--	--	139.5	120	15	--	--	144.5	71.5	57.333
PB	18	--	--	390.75	378.6	56.667	--	--	141.5	85.4	6.6667
PB	22.1	--	--	111	5	0	--	--	32.5	52	7
PB	20.1	--	--	257	23	13	--	--	86	15	4
PB	24.1	--	--	181	19	33	--	--	62	15	40
PB	24.2	--	--	334	236	32	--	--	194	21	31
PB	20.3	--	--	188	254	0	--	--	37	63	190
PB	2	--	--	--	31	3.5	--	--	--	4.5	11.5
PT	0	11	32	--	--	--	35	14	--	--	--
PT	2	90	0	42.5	22.5	0	12	28	112.5	112	63.5
PT	7	37	0	--	--	--	79	103	--	--	--
PT	15	4	23	--	--	--	10	59	--	--	--
PT	18	31	133	76.25	81.4	8.6667	52	129	269	478.2	383.33
PT	21	14	259	--	--	--	47	116	--	--	--
PT	24	0	176	202	164	45	74	243	200	614	521
PT	8	--	--	27	15.5	3	--	--	458	1209.5	1242
PT	14	--	--	150	140.67	5.6667	--	--	256.5	209	92
PT	16	--	--	35.5	55.5	5.3333	--	--	181.5	107	54
PT	20	--	--	91	21	0	--	--	254	250	326

Appendix B (continued).

Habitat Unit	Chinook					Steelhead >1					
	'93.1	'93.2	'94.1	'94.2	'94.3	'93.1	'93.2	'94.1	'94.2	'94.3	
IPR	22	--	--	42.5	0	0	--	--	150	1	0
IPR	24	--	--	52	24	9	--	--	132	28	2
IPR	20.1	--	--	154	18	0	--	--	137	26	0
IPR	20.2	--	--	77	17	0	--	--	206	150	7
RiMa	7	3	1	5	0	0	20	10	24.5	27.5	18
RiMa	17	4.5	4	22	2	1	14.5	18	36.5	48.5	61
RiMa	27	27	5	18	1.5	0	12	7	37.5	3	4
RiMa	37	16	12	35	7	4	36	29	69	80	62
RiMa	47	0	0	0	0	0	2	1	1.5	0	1
RiMa	57	24	3	25.5	14	1	15	17	51	60	20
RiMa	67	0	0	2	0	0	8	1	11	20	14
RiMa	73	2	0	7	5	0	14	13	20	21	9
RiMa	83	0	0	0	1	0	9	3	11	8	3
RiMa	92	0	0	0	0	0	11	11	14	28.5	9
RiMi	1	0	0	0	0	0	0	2	10	11	4
RiMi	8	0	0	1.3333	0	0	14	44	58.667	180	197
RiMi	15	0	0	1	11	0	16	40	41	78	89
RiMi	22	0	0	0	0	0	6	0	16.5	8	15
RiMi	25	--	--	0	0	0	--	--	46	119	96
RiMi	29	0	0	0	0	0	16	11.25	23	10	32
RiMi	34	0	1	1	6	0	16.5	9.25	46	87	66
RiMi	37	--	--	--	0	0	--	--	--	8	3
RiMi	41	0	0	0	0	0	7.5	2.5	0	10	3
RiMi	45	--	--	0	0	0	--	--	8	12	13
RuMa	4	9	6	15.5	9	--	8	17	60	39	--
RuMa	9	73	1	51.5	1.5	--	35	5	50	15	--
RuMa	14	4	0	26	0	--	4	7.5	12	1.5	--
RuMa	19	26	4	33.5	10	--	35	26	70.5	31	--
RuMa	24	17	11	26.5	7	--	13	7	47	22.5	--
RuMa	29	13	1	45.5	5.5	--	10.5	21	73.5	23.5	--
RuMa	34	43	14.5	22	6.5	--	20	18	42	36	--
RuMa	42	28	19.5	56	25.5	--	21.5	14	24.5	24	--
RuMa	47	15	4	11.5	0	--	16	20	36.5	16	--
RuMa	52	1	1	5.5	10	--	22	22	40.5	13	--
RuMi	1	--	--	42.667	9	--	--	--	58.667	164	--
RuMi	3	2	0	8	0	--	36	47	109.33	97.5	--
RuMi	5	--	--	0	9	--	--	--	55	43.5	--
RuMi	7	12	8	20	6	--	22	24.33	64	7.5	--
RuMi	11	3.75	1.25	22	5	--	27.5	23.75	168	108	--
RuMi	13	0	0	1	0	--	6	13	25	53	--
RuMi	15	--	--	0	0	--	--	--	70	145	--
RuMi	17	2	4	15	2	--	25	40	70	117	--
RuMi	21	20	21	23	55.5	--	22	30	68	109.5	--
RuMi	25	2	0	0	2	--	30	42	72.5	98	--

Appendix C. Paired Counts of juvenile chinook (CK) and steelhead >1 (SH) in the Middle Fork Smith River, California with coefficient of variation (CV=standard deviation/mean). Habitat types are coded as follows: (1) pool head, (2) pool body, (3) pool tail, (4) margin.

UNIT	TIME	HABITAT	CK1	CK2	CV	SH1	SH2	CV
PH14	93.1	1	60	72	0.090909	21	23	0.045455
PH14	93.1	1	63	45	0.166667	39	27	0.181818
PH10	93.1	1	162	200	0.104972	14	12	0.076923
PH6	93.1	1	25	27	0.038462	17	19	0.055556
PH6	93.1	1	86	77	0.055215	49	49	0
PH6	93.1	1	79	65	0.097222	48	51	0.030303
IM17	93.1	4	4	5	0.111111	16	13	0.103448
RM42	93.1	4	15	14	0.034483	17	19	0.055556
RM34	93.1	4	1	1	0	22	20	0.047619
RM42	93.1	4	21	18	0.076923	14	14	0
RM42	93.1	4	32	17	0.306122	41	28	0.188406
RM34	93.2	4	46	40	0.069767	23	17	0.15
RM42	93.2	4	28	28	0	20	23	0.069767
RM47	93.2	4	14	16	0.066667	16	16	0
RM29	93.2	4	10	14	0.166667	13	13	0
RM24	93.2	4	2	6	0.5	17	19	0.055556
RM9	93.2	4	12	14	0.076923	10	11	0.047619
RM42	94.1	4	12	11	0.043478	12	8	0.2
RM1	94.2	4	86	76	0.061728	13	7	0.3
Average CV:					0.108806			0.084633

Appendix D. Repeat counts of juvenile chinook (CK) and steelhead >1 in the Middle Fork Smith River, California with coefficient of variation (CV=standard deviation/mean). Habitat types are coded as follows: (1) pool head, (2) pool body, (3) pool tail, (4) margin. Counts for margin units in 1994 were lost after CVs were calculated. Averages at bottom exclude counts from pool units and margins in 1994.
(continued on next page).

UNIT	TIME	HAB	CK	rptCK	CV	SH	rptSH	CV
RM24	93.1	4	17	17	0	13	8	0.238095
RM47	93.1	4	15	13	0.071429	16	8	0.333333
PT14	93.1	3	31	31	0	52	52	0
IM73	93.1	4	2	3	0.2	14	12	0.076923
IM37	93.1	4	15	21	0.166667	45	42	0.034483
R13	93.1	4	0	0		3	3	0
I22	93.1	4	0	0		1.5	2	0.142857
IM27	93.1	4	27	29	0.035714	12	21	0.272727
RM19	93.1	4	14	13	0.037037	17	17	0
PT11	93.1	3	20	18	0.052632	86	85	0.005848
PH16	93.1	1	414	349	0.08519	106	95	0.054726
PB16	93.1	2	317	292	0.041051	71	62	0.067669
PH18	93.1	1	342	355	0.018651	74	75	0.006711
PH6	93.1	1	150	161	0.03537	59.5	66	0.051793
PB12	93.1	2	28	31	0.050847	18	26	0.181818
PB16	93.1	2	218	225	0.015801	39	16	0.418182
PT14	93.1	3	154	161	0.022222	139	144	0.017668
PB12	93.1	2	149	163	0.044872	26	32	0.103448
PB8	93.1	2	60	116	0.318182	446	637	0.176362
RM42	93.2	4	29	30	0.016949	22	23	0.022222
RM19	93.2	4	27	24	0.058824	41	41	0
IM27	93.2	4	5	5	0	7	7	0
PT05	93.2	3	32	27	0.084746	79	95	0.091954
PT14	93.2	3	133	97	0.156522	129	83	0.216981
IM67	93.2	4	0	0		6	6	0
IM37	93.2	4	22	23	0.022222	40	41	0.012346
RM14	93.2	4	0	0		7.5	7	0.034483
RM19	93.2	4	4	3	0.142857	26	27	0.018868
IM37	93.2	4	12	12	0	29	28	0.017544
IM57	93.2	4	3	4	0.142857	17	18	0.028571
RM42	93.2	4	19.5	26	0.142857	14	18	0.125
PH20	93.2	1	530	587	0.05103	81	90	0.052632
PH14	93.2	1	553	657	0.08595	257	264	0.013436
PH20	93.2	1	472	437	0.038504	223	187	0.087805
PH14	93.2	1	276	315	0.06599	285	313	0.046823
PH16	93.2	1	131	121	0.039683	136	135	0.00369
PH2	93.2	1	89	115	0.127451	36	39	0.04
PB16	93.2	2	409	404	0.00615	88	74	0.08642
PT5	93.2	3	0	0		103	111	0.037383
PT17	93.2	3	259	229	0.061475	116	167	0.180212
PH24	93.2	1	126	128	0.007874	144	141	0.010526
PH26	93.2	1	108	116	0.035714	135	148	0.045936
PB20	93.2	2	368	349	0.026499	24	50	0.351351

Appendix D (continued).

UNIT	TIME	HAB	CK	rptCK	CV	SH	rptSH	CV
2	94.1	pool	302	369	0.141211	295	323	0.064074
14	94.1	pool	841	808	0.028301	568	761	0.205375
16	94.1	pool	871	685	0.169051	738	529	0.233284
18	94.1	pool	1024	994	0.021024	683	621	0.06724
22	94.1	pool	665	592	0.08213	385	425	0.069838
2	94.2	pool	104	102	0.01373	329	285	0.101344
8	94.2	pool	107	79	0.212892	2103	2443	0.10577
12	94.2	pool	305	304	0.002322	316	282	0.080407
14	94.2	pool	350	454	0.182933	419	489	0.109025
16	94.2	pool	717	636	0.084665	570	592	0.026775
18	94.2	pool	1023	1244	0.137866	981	1144	0.108478
22	94.2	pool	427	476	0.07674	749	835	0.076782
2	94.3	pool	9	14	0.307438	206	261	0.166556
14	94.3	pool	122	186	0.293863	365	353	0.023636
16	94.3	pool	96	102	0.042855	431	493	0.094893
18	94.3	pool	127	147	0.103227	606	642	0.040795
	94.1	4			0.147754			0.01003
	94.1	4			0.240149			0.120359
	94.1	4			0.108786			0.235702
	94.1	4			0.593057			0.02357
	94.1	4			0.041191			0.028284
	94.1	4			0.184463			0.096864
	94.1	4			0.385695			0.157135
	94.1	4			0.075761			0.144308
	94.1	4			0.015541			0.105826
	94.1	4			0.157135			0.24513
	94.1	4			0			0.471405
	94.1	4			0.194108			0.02773
	94.1	4			0			0.385695
	94.1	4			0			0.101015
	94.1	4			0.064282			0.058118
	94.1	4			0			0.144308

Appendix D (continued).

UNIT	TIME	HAB	CK	rptCK	CV	SH	rptSH	CV
22	94.1	2	119	103	0.0720721	29	36	0.1076923
16	94.1	2	129	150	0.0752688	158	131	0.0934256
14	94.1	2	231	219	0.0266667	51	74	0.184
14	94.1	3	192	108	0.28	227	286	0.1150097
16	94.1	3	47	24	0.3239437	203	160	0.1184573
18	94.1	3	209	19	0.8333333	318	257	0.106087
2	94.1	3	40	45	0.0588235	101	124	0.1022222
2	94.1	1	262	324	0.105802	194	199	0.0127226
14	94.1	1	418	481	0.0700779	290	401	0.1606368
18	94.1	1	648	648	0	312	219	0.1751412
16	94.1	1	695	511	0.1525705	377	238	0.2260163
18	94.1	2	167	327	0.3238866	53	145	0.4646465
10	94.1	1	497	584	0.080481	283	305	0.037415
22	94.1	1	494	456	0.04	196	249	0.1191011
	94.2	4			0.0277297			0.4714045
	94.2	4						
	94.2	4						
	94.2	4			1.4142136			0.1767767
	94.2	4			0			0.1240538
	94.2	4						
	94.2	4			0.2828427			0.269374
	94.2	4			0			1.4142136
	94.2	4			0.7071068			0.2737188
	94.2	4			0.3142697			0.7977615
	94.2	4			1.4142136			0.0942809
	94.2	4			0.3030458			0.3064129
	94.2	4			0.404061			0.2828427
	94.2	4			0			0.3856946
	94.2	4			0.326357			0.274986
	94.2	4			0.3856946			0.0300897
	94.2	4			0.404061			0.265165
	94.2	4						
	94.2	4			1.4142136			0.0728976
	94.2	4			0.4714045			0.4714045
22	94.2	2	2	8	0.6	86	18	0.6538462
12	94.2	2	38	0	1	0	0	0
16	94.2	2	101	139	0.1583333	34	109	0.5244755
18	94.2	2	429	525	0.1006289	126	100	0.1150442
12	94.2	2	71	94	0.1393939	37	38	0.0133333
8	94.2	3	21	10	0.3548387	1200	1219	0.0078545
2	94.2	2	41	21	0.3225806	4	5	0.1111111
8	94.2	2	48	46	0.0212766	334	608	0.2908705
16	94.2	1	553	449	0.1037924	401	404	0.0037267
18	94.2	1	540	662	0.1014975	425	440	0.017341
8	94.2	1	38	23	0.2459016	569	616	0.0396624
2	94.2	1	36	63	0.2727273	208	173	0.0918635
12	94.2	1	10	32	0.5238095	1	2	0.3333333
14	94.2	1	145	265	0.2926829	210	275	0.1340206
22	94.2	1	425	468	0.0481523	663	815	0.1028417
12	94.2	1	191	173	0.0494505	205	181	0.0621762
2	94.2	3	27	18	0.2	117	107	0.0446429
16	94.2	3	63	48	0.1351351	135	79	0.2616822
18	94.2	3	54	57	0.027027	430	604	0.1682785
14	94.2	3	107	84	0.1204188	196	183	0.0343008
16	94.3	3	3	4	0.1428571	56	54	0.0181818
16	94.3	2	17	18	0.0285714	66	67	0.0075188
14	94.3	2	0	12	1	14	56	0.6
2	94.3	3	0	0	0	49	78	0.2283465
18	94.3	3	8	7	0.0666667	407	397	0.0124378
14	94.3	3	2	13	0.7333333	78	90	0.0714286
18	94.3	2	94	60	0.2207792	1	12	0.8461538
16	94.3	1	76	80	0.025641	309	372	0.092511
2	94.3	1	6	10	0.25	148	169	0.0662461
14	94.3	1	120	161	0.1459075	273	207	0.1375
2	94.3	2	3	4	0.1428571	9	14	0.2173913
10	94.3	1	14	13	0.037037	304	367	0.0938897
18	94.3	1	25	80	0.5238095	191	233	0.0990566
Average:					0.1514433			0.1281693

Appendix E. Revisit counts of juvenile chinook (CK) and steelhead >1 in the Middle Fork Smith River California with coefficient of variation (CV=standard deviation/mean). Habitat types are coded as follows: (1) pool head, (2) pool body, (3) pool tail, (4) margin. Averages at bottom exclude counts for pool units and margins in 1994.

UNIT	TIME	HAB	CK	rvstCK	CV	SH	rvstSH	CV	
PH14	93.1	1	274	456	0.2493	184	255	0.1617	
PH20	93.1	1	512	530	0.0173	114	81	0.1692	
PH14	93.1	1	456	553	0.0961	255	257	0.0039	
PB20	93.1	2	294	368	0.1118	21	24	0.0667	
PT14	93.1	3	31	61	0.3261	52	49	0.0297	
PT5	93.1	3	37	32	0.0725	79	79	0.0000	
IM37	93.1	4	16	22	0.1579	36	40	0.0526	
IM73	93.1	4	2	5	0.4286	14	8	0.2727	
IM67	93.1	4	0	0		8	6	0.1429	
RM19	93.1	4	26	27	0.0189	35	41	0.0789	
RM24	93.1	4	17	19	0.0556	13	12	0.0400	
RM47	93.1	4	15	15	0.0000	16	20	0.1111	
RM42	93.1	4	28	29	0.0175	21.5	22	0.0115	
PH8	93.2	1	85	105	0.1053	236	276	0.0781	
PH14	93.2	1	276	332	0.0921	285	306	0.0355	
PH18	93.2	1	346	342	0.0058	84	74	0.0633	
PB12	93.2	2	25	28	0.0566	20	18	0.0526	
PB16	93.2	2	409	317	0.1267	88	71	0.1069	
PB8	93.2	2	128	60	0.3617	508	446	0.0650	
PT2	93.2	3	0	1	1.0000	28	30	0.0345	
PT14	93.2	3	133	154	0.0732	129	139	0.0373	
PT11	93.2	3	23	20	0.0698	59	86	0.1862	
PT5	93.2	3	0	6	1.0000	103	302	0.4914	
IM37	93.2	4	12	15	0.1111	29	45	0.2162	
IM73	93.2	4	0	1	1.0000	13	9	0.1818	
IM67	93.2	4	0	0		1	2	0.3333	
RM19	93.2	4	4	14	0.5556	26	17	0.2093	
RM47	93.2	4	4	3	0.1429	20	18	0.0526	
RM42	93.2	4	19.5	11.5	0.2581	14	10	0.1667	
RM24	93.2	4	11	12	0.0435	7	13	0.3000	
	18	94.1	1	648	674	0.0197	312	245	0.1203
	12	94.1	1	397	277	0.1780	219	170	0.1260
	12	94.1	2	419	309	0.1511	79	134	0.2582
	18	94.1	2	167	668	0.6000	53	231	0.6268
	18	94.1	3	209	42	0.6653	318	277	0.0689
	14	94.2	1	482	145	0.5375	337	210	0.2322
	18	94.2	1	696	388	0.2841	593	327	0.2891
	18	94.2	1	540	696	0.1262	425	593	0.1650
	12	94.2	1	191	10	0.9005	205	1	0.9903
	14	94.2	2	485	98	0.6638	63	13	0.6579
	18	94.2	2	328	284	0.0719	53	59	0.0536
	18	94.2	2	429	328	0.1334	126	53	0.4078
	12	94.2	2	71	38	0.3028	37	0	1.0000
	14	94.2	3	231	107	0.3669	248	196	0.1171
	18	94.2	3	54	74	0.1563	430	464	0.0380
	18	94.2	3	74	60	0.1045	464	325	0.1762
	18	94.3	1	114	25	0.6403	212	191	0.0521
	16	94.3	1	84	76	0.0500	329	309	0.0313
	14	94.3	1	69	120	0.2698	244	273	0.0561
	14	94.3	2	5	0	1.0000	35	14	0.4286
	16	94.3	2	10	17	0.2593	39	66	0.2571
	18	94.3	2	16	94	0.7091	7	1	0.7500
	14	94.3	3	2	2	0.0000	108	78	0.1613
	18	94.3	3	11	8	0.1579	346	407	0.0810
	16	94.3	3	9	3	0.5000	52	56	0.0370
	12	94.1 pool		1040	664	0.3121	360	366	0.0117
	18	94.1 pool		1024	1384	0.2114	683	753	0.0689
	14	94.2 pool		1197	350	0.7743	648	419	0.3035
	18	94.2 pool		1023	1098	0.0500	981	1110	0.0872
	18	94.2 pool		1036	732	0.2432	1172	711	0.3462
	14	94.3 pool		76	122	0.3286	387	365	0.0414
	16	94.3 pool		103	96	0.0497	420	431	0.0183
Average:					0.2906			0.1988	

Appendix F. Repeated counts and estimates of juvenile chinook (CK) and steelhead >1 abundance in habitat units of the Middle Fork Smith River, California by two methods. \hat{y}_i' is the average of original count (CK) and repeat count 10 min. - 1 hr. later (CKrpt or the average of a revisit count 1-4 days later (CKrvst) and repeat count 10 min. - 1 hr. after revisit count (CKrpt2). One of the preceding pairs plus the revisit count or plus the original count, respectively, are averaged for \hat{y}_i'' . Rows are listed by time and habitat type: (1) pool head, (2) pool body, (3) pool tail, and (4) margin (Continued on next page for steelhead with similar notation).

UNIT	HABITAT	TIME	CK	CKrpt	CKrvst	CKrpt2	\hat{y}_i'	\hat{y}_i''	V(\hat{y}_i')	V(\hat{y}_i'')
1114	1	93.1	274		553	657	605.0	494.7	5408.0	39224.3
1120	1	93.1	512		530	587	558.5	543.0	1624.5	1533.0
1220	2	93.1	294		368	349	358.5	337.0	180.5	1477.0
1305	3	93.1	37		32	27	29.5	32.0	12.5	25.0
1314	3	93.1	31	31	61		31.0	41.0	0.0	300.0
2137	4	93.1	16		22	23	22.5	20.3	0.5	14.3
2173	4	93.1	2	3	5		2.5	3.3	0.5	2.3
3119	4	93.1	26		27	24	25.5	25.7	4.5	2.3
3124	4	93.1	17	17	19		17.0	17.7	0.0	1.3
3142	4	93.1	28		29	30	29.5	29.0	0.5	1.0
3147	4	93.1	15	13	15		14.0	14.3	2.0	1.3
1114	1	93.2	276	315	332		295.5	307.7	760.5	824.3
1118	1	93.2	346		342	355	348.5	347.7	84.5	44.3
1208	2	93.2	128		60	116	88.0	101.3	1568.0	1317.3
1209	2	93.2	25		28	31	29.5	28.0	4.5	9.0
1216	2	93.2	409	404	317	292	406.5	376.7	12.5	2676.3
1311	3	93.2	23		20	18	19.0	20.3	2.0	6.3
1314	3	93.2	133	97	154	161	115.0	128.0	648.0	831.0
2137	4	93.2	12	12	15	21	12.0	13.0	0.0	3.0
3119	4	93.2	4	3	14	13	3.5	7.0	0.5	37.0
3142	4	93.2	21	25	12		23.0	19.3	8.0	44.3
18	1	94.1	648	648	674		648.0	656.7	0.0	225.3
18	2	94.1	167	327	668		247.0	387.3	12800.0	65480.3
18	3	94.1	209	19	42		114.0	90.0	18050.0	10753.0
12	1	94.2	191	173	10	32	182.0	124.7	162.0	9942.3
14	1	94.2	482		145	265	205.0	297.3	7200.0	29176.3
18	1	94.2	540	662	696		601.0	632.7	7442.0	6729.3
12	2	94.2	71	94	38	0	82.5	67.7	264.5	792.3
14	2	94.2	485		98	105	101.5	229.3	24.5	49036.3
18	2	94.2	429	525	328		477.0	427.3	4608.0	9704.3
14	3	94.2	231		107	84	95.5	140.7	264.5	6252.3
18	3	94.2	54	57	74		55.5	61.7	4.5	116.3
14	1	94.3	69		120	161	140.5	116.7	840.5	2124.3
16	1	94.3	84		76	80	78.0	80.0	8.0	16.0
18	1	94.3	114		25	80	52.5	73.0	1512.5	2017.0
14	2	94.3	5		0	12	6.0	5.7	72.0	36.3
16	2	94.3	10		17	18	17.5	15.0	0.5	19.0
18	2	94.3	16		94	60	77.0	56.7	578.0	1529.3
14	3	94.3	2		2	13	7.5	5.7	60.5	40.3
16	3	94.3	9		3	4	3.5	5.3	0.5	10.3
18	3	94.3	11		8	7	7.5	8.7	0.5	4.3

Appendix F (continued).

UNIT	HABITAT	TIME	SH	SHrpt	SHrvst	SHrpt2	yi'	yi"	V(yi')	V(yi")
1114	1	93.1	184		257	264	260.5	235	24.5	1963
1120	1	93.1	114		81	90	85.5	95	40.5	291
1220	2	93.1	21		24	50	21	22.5	338	254.33333
1305	3	93.1	79		79	95	87	84.333333	128	85.333333
1314	3	93.1	52	52	49		52	51	0	3
2137	4	93.1	36		40	41	40.5	39	0.5	7
2167	4	93.1	8		6	6	6	6.6666667	0	1.3333333
2173	4	93.1	14	12	8		13	11.333333	2	9.3333333
3119	4	93.1	35		41	41	41	39	0	12
3124	4	93.1	13	8	12		10.5	11	12.5	7
3142	4	93.1	20		22	23	22.5	21.666667	0.5	2.3333333
3147	4	93.1	16	8	20		12	14.666667	32	37.333333
1114	1	93.2	285	313	306		299	301.33333	392	212.33333
1118	1	93.2	84		74	75	74.5	77.666667	0.5	30.333333
1208	2	93.2	508		446	637	541.5	530.33333	18240.5	9494.3333
1209	2	93.2	20		18	26	22	21.333333	32	17.333333
1216	2	93.2	88	74	71	62	81	77.666667	98	82.333333
1305	3	93.2	103	111	302		107	172	32	12691
1311	3	93.2	59		86	85	85.5	76.666667	0.5	234.33333
1314	3	93.2	129	83	139	144	106	117	1058	892
2137	4	93.2	29	28	45	42	28.5	34	0.5	91
3119	4	93.2	26	27	17	17	26.5	23.333333	0.5	30.333333
3142	4	93.2	14	17	12		15.5	14.333333	4.5	6.3333333
18	1	94.1	312	219	245		265.5	258.66667	4324.5	2302.3333
18	2	94.1	53	145	231		99	143	4232	7924
18	3	94.1	318	257	277		287.5	284	1860.5	967
12	1	94.2	205	181	1	2	193	129	288	12432
14	1	94.2	337		210	275	242.5	274	2112.5	4033
18	1	94.2	425	440	593		432.5	486	112.5	8643
12	2	94.2	37	38	0	0	37.5	25	0.5	469
14	2	94.2	63		13	23	18	33	50	700
18	2	94.2	126	100	53		113	93	338	1369
14	3	94.2	248		196	183	189.5	209	84.5	1183
18	3	94.2	430	604	464		517	499.33333	15138	8505.3333
14	1	94.3	244		273	207	240	241.33333	2178	1094.3333
16	1	94.3	329		309	372	340.5	336.66667	1984.5	1036.3333
18	1	94.3	212		191	233	212	212	882	441
14	2	94.3	35		14	56	35	35	882	441
16	2	94.3	39		66	67	39	52.5	0.5	252.33333
18	2	94.3	7		1	12	6.5	6.6666667	60.5	30.333333
14	3	94.3	108		78	90	84	92	72	228
16	3	94.3	52		56	54	55	54	2	4
18	3	94.3	346		407	397	402	383.33333	50	1070.3333

Appendix G. Actual catches, measured flows, and estimated total emigrants at Site 12 km on the Middle Fork Smith River, California during 1994. Flow through nets (cms) was estimated using surface velocity on dates prior to July 14. Only one net was fished on July 3. Traps on dates preceding June 27 were set deeper than 0.1 m below water surface (ws).

Date	Height (m)	Q (cms)	Q (cms)	Q (cms)	Q (cms)	Q (cms)	Actual Catch			Estimated Total		
	Above ws	Left Net	Right Net	Both Nets	Middle Fk	Ratio	Chinook	Sthd <1	Sthd >1	Chinook	Sthd <1	Sthd >1
27-Jun	-0.11	na	na	2.29	7.72	3.37	97	0	26	326.7	0.0	87.6
28-Jun	-0.09	na	na	2.29	7.57	3.30	148	1	25	488.2	3.3	82.5
29-Jun	-0.08	na	na	1.02	7.50	7.36	151	0	16	1110.7	0.0	117.7
30-Jun	0.05	na	na	1.13	7.34	6.48	189	3	17	1224.7	19.4	110.2
03-Jul	-0.05	na	na	0.48	7.26	15.08	71	4	3	1070.8	60.3	45.2
09-Jul	0.20	na	na	1.16	6.16	5.31	523	13	33	2775.7	69.0	175.1
12-Jul	0.00	na	na	1.37	5.83	4.27	183	13	16	780.5	55.4	68.2
13-Jul	0.00	na	na	1.37	5.70	4.17	114	15	12	474.9	62.5	50.0
14-Jul	0.24	0.71	0.68	1.39	5.56	4.00	188	14	23	752.8	56.1	92.1
15-Jul	0.41	0.54	0.58	1.12	5.49	4.91	204	22	15	1000.7	107.9	73.6
16-Jul	0.30	0.58	0.65	1.22	5.37	4.39	182	14	11	799.7	61.5	48.3
17-Jul	0.40	0.52	0.57	1.09	5.30	4.85	168	7	8	814.1	33.9	38.8
18-Jul	0.40	0.53	0.52	1.06	5.18	4.90	207	10	6	1014.3	49.0	29.4
19-Jul	0.40	0.55	0.53	1.07	5.12	4.78	180	3	4	860.2	14.3	19.1
20-Jul	0.27	0.58	0.61	1.19	5.12	4.30	103	8	2	443.0	34.4	8.6
21-Jul	0.40	0.42	0.51	0.93	5.05	5.40	59	7	2	318.9	37.8	10.8
22-Jul	0.30	0.54	0.54	1.08	5.12	4.74	65	7	1	308.3	33.2	4.7
26-Jul	0.30	0.55	0.52	1.07	4.87	4.56	148	12	9	675.6	54.8	41.1
27-Jul	0.30	0.57	0.58	1.16	4.81	4.17	90	18	4	375.1	75.0	16.7
28-Jul	0.37	0.53	0.50	1.03	4.75	4.62	80	6	9	369.4	27.7	41.6
29-Jul	0.27	0.51	0.50	1.01	4.75	4.72	91	11	15	429.5	51.9	70.8
30-Jul	0.30	0.49	0.50	0.98	4.68	4.76	141	25	14	670.7	118.9	66.6
31-Jul	0.34	0.49	0.47	0.96	4.62	4.80	99		10	475.1		48.0
01-Aug	0.30	0.48	0.48	0.96	4.55	4.74	135		10	639.7		47.4
02-Aug	0.30	0.52	0.54	1.06	4.43	4.16	91	23	10	378.5	95.7	41.6
03-Aug	0.12	0.73	0.73	1.45	4.36	3.00	249	65	36	746.6	194.9	107.9
04-Aug	0.12	0.77	0.70	1.48	4.36	2.95	119		11	351.6		32.5
05-Aug	0.18	0.62	0.68	1.30	4.30	3.31	101		35	334.5		115.9
09-Aug	0.21	0.63	0.60	1.23	4.18	3.39	43		8	145.7		27.1
10-Aug	0.18	0.69	0.66	1.35	4.11	3.04	72		12	219.1		36.5
11-Aug	0.15	0.73	0.68	1.42	4.05	2.86	62		16	177.6		45.8
12-Aug	0.15	0.66	0.64	1.30	4.00	3.07	45	23	8	138.2	70.7	24.6
13-Aug	0.12	0.57	0.60	1.17	4.00	3.43	24	29	10	82.2	99.3	34.3
14-Aug	0.15	0.67	0.52	1.19	4.00	3.36	19		7	63.8		23.5
15-Aug	0.24	0.60	0.56	1.15	4.00	3.47	30	22	13	104.1	76.3	45.1
17-Aug	0.21	0.58	0.54	1.12	3.89	3.47	9		6	31.2		20.8
18-Aug	0.21	0.55	0.59	1.13	3.82	3.37	21		3	70.8		10.1
24-Aug	0.21	0.55	0.45	1.00	3.65	3.63	12	1	0	43.6	3.6	0.0
25-Aug	0.15	0.68	0.58	1.27	3.60	2.85	29	2	5	82.6	5.7	14.2
26-Aug	0.15	0.63	0.75	1.38	3.60	2.61	32	0	4	83.6	0.0	10.5
27-Aug	0.18	0.58	0.64	1.22	3.55	2.90	27	4	3	78.3	11.6	8.7
30-Aug	0.18	0.39	0.60	0.99	3.43	3.45	29	5	2	100.1	17.3	6.9
14-Sep	0.15	0.61	0.71	1.32	na	na	16	6	7	na	na	na
20-Sep	0.15	0.59	0.62	1.21	na	na	4		2	na	na	na
27-Sep	0.15	0.82	0.68	1.50	na	na	9		3	na	na	na
13-Oct	na	na	na	na	na	na	32		10	na	na	na

Appendix H. Summary statistics for fork length of steelhead >1 trapped at two sites on the Middle Fork Smith River, California during the summers of 1993 and 1994. (Continued).

Site	Date	Count	Fork Length (mm) of Steelhead 1+			
			Minimum	Maximum	Mean	Std. Deviation
0 km	03-Jul 93	1	116	116	116	
0 km	16-Jul 93	6	91	143	110	17.73
0 km	29-Jul 93	2	128	137	132.5	6.36
0 km	30-Jul 93	2	119	125	122	4.24
0 km	31-Jul 93	7	94	135	108	14.5
0 km	06-Aug 93	5	126	152	139.8	10.94
0 km	12-Aug 93	6	101	126	113.33	10.41
0 km	18-Aug 93	1	160	160	160	
12 km	29-Jun 93	4	98	126	112.75	12.31
12 km	08-Jul 93	5	110	171	134.2	27.78
12 km	13-Jul 93	3	111	134	123.33	11.59
12 km	28-Jul 93	2	119	160	139.5	28.99
12 km	03-Aug 93	5	108	130	119.6	9.24
12 km	10-Aug 93	23	103	156	131.13	13.68
12 km	17-Aug 93	16	110	141	127.75	8.1
0 km	20-Jul 94	3	120	135	127.67	7.51
0 km	28-Jul 94	4	119	155	134.5	17.9
0 km	25-Aug 94	10	114	157	132.3	14.27
0 km	31-Aug 94	3	121	143	130.33	11.37
0 km	09-Sep 94	3	114	145	132	16.09
0 km	14-Sep 94	2	125	147	136	15.56
0 km	11-Oct 94	1	179	179	179	
12 km	03-Jun 94	7	98	159	126.14	19.84
12 km	04-Jun 94	7	87	155	122.43	21.35
12 km	05-Jun 94	4	81	139	114.25	26.92
12 km	06-Jun 94	8	95	136	120.13	12.03
12 km	08-Jun 94	1	105	105	105	
12 km	09-Jun 94	3	126	135	129	5.2
12 km	10-Jun 94	11	90	142	115.36	14.95
12 km	11-Jun 94	7	101	124	114.14	9.48
12 km	12-Jun 94	12	108	148	126.58	11.56
12 km	13-Jun 94	11	99	147	122.91	17.75
12 km	14-Jun 94	5	109	133	119.8	10.89
12 km	15-Jun 94	4	105	125	114.5	8.54
12 km	16-Jun 94	7	98	141	126.14	14.59
12 km	17-Jun 94	28	96	145	115.75	12.39
12 km	18-Jun 94	18	99	144	121.83	10.05
12 km	19-Jun 94	18	99	144	121.83	10.05
12 km	20-Jun 94	4	115	142	126.75	11.67
12 km	21-Jun 94	18	100	143	125.17	12.24
12 km	22-Jun 94	5	114	141	128.6	10.36
12 km	23-Jun 94	5	106	162	129.4	26.01
12 km	24-Jun 94	4	103	145	124.25	19.96
12 km	25-Jun 94	14	106	145	122	11.9
12 km	27-Jun 94	26	97	136	115.65	10.7
12 km	28-Jun 94	25	94	158	116.64	15.85
12 km	29-Jun 94	15	95	158	120.2	16.37
12 km	30-Jun 94	18	94	144	121.89	12.85

Appendix H (Continued).

Site	Date	Count	Fork Length (mm) of Steelhead 1+			Std. Deviation
			Minimum	Maximum	Mean	
12 km	01-Jul 94	11	91	142	109.82	13.98
12 km	02-Jul 94	7	102	132	113.43	9.76
12 km	03-Jul 94	2	97	124	110.5	19.09
12 km	07-Jul 94	12	90	136	118.42	12.3
12 km	08-Jul 94	9	99	142	118	16.51
12 km	09-Jul 94	34	90	153	118.32	15.14
12 km	10-Jul 94	21	92	159	121.67	15.94
12 km	11-Jul 94	23	95	152	120.52	13.73
12 km	12-Jul 94	16	98	139	114.38	10.71
12 km	13-Jul 94	12	100	140	120.75	14.15
12 km	14-Jul 94	23	91	139	118.3	13.81
12 km	15-Jul 94	14	94	147	124.86	13.36
12 km	16-Jul 94	11	104	156	125.45	13.72
12 km	17-Jul 94	8	105	145	120.88	12.48
12 km	18-Jul 94	6	109	132	121.33	9.29
12 km	19-Jul 94	6	107	136	124	10.55
12 km	21-Jul 94	8	94	162	126.88	19.71
12 km	22-Jul 94	7	106	145	125.57	14.71
12 km	26-Jul 94	11	105	140	124.09	11.18
12 km	27-Jul 94	19	97	161	123.47	16.22
12 km	28-Jul 94	6	100	132	116.83	11.57
12 km	29-Jul 94	15	108	135	120.67	7.5
12 km	30-Jul 94	12	99	160	129.17	17.11
12 km	31-Jul 94	10	92	152	123.6	17.87
12 km	01-Aug 94	9	84	138	118	15.16
12 km	02-Aug 94	10	100	167	127.6	23.41
12 km	03-Aug 94	36	105	161	124.31	11.05
12 km	09-Aug 94	6	113	131	122.83	8.28
12 km	10-Aug 94	11	105	141	126.18	11.15
12 km	11-Aug 94	16	109	149	124.88	11.73
12 km	12-Aug 94	8	106	143	123.63	11.95
12 km	15-Aug 94	13	113	149	128.15	11.23
12 km	26-Aug 94	4	123	134	129.25	4.57
12 km	27-Aug 94	3	103	136	125	19.05
12 km	30-Aug 94	2	126	163	144.5	26.16
12 km	11-Sep 94	7	120	143	131.43	7.46
12 km	20-Sep 94	2	125	160	142.5	24.75
12 km	27-Sep 94	3	123	128	125.33	2.52
12 km	13-Oct 94	10	140	172	153.9	10.8

Appendix I. Summary statistics for fork length of chinook trapped at two sites on the Middle Fork Smith River, California during the summers of 1993 and 1994. (Continued).

Site	Date	Count	Fork Length (mm) of Chinook			
			Minimum	Maximum	Mean	Std. Deviation
0 km	03-Jul 93	30	55	81	68.67	8.52
0 km	16-Jul 93	270	48	94	70.05	10.2
0 km	29-Jul 93	34	56	91	75.41	5.23
0 km	30-Jul 93	61	58	94	77.05	8.27
0 km	31-Jul 93	66	57	95	77.86	9.13
0 km	06-Aug 93	54	69	94	80.89	7.32
0 km	12-Aug 93	92	59	100	81.39	9.15
0 km	18-Aug 93	41	67	94	79.68	6.54
12 km	29-Jun 93	22	62	90	74.73	7.9
12 km	08-Jul 93	15	62	90	74.27	8.6
12 km	13-Jul 93	32	62	85	75.75	8.37
12 km	15-Jul 93	5	70	95	80.6	9.96
12 km	28-Jul 93	287	57	105	84.1	9.16
12 km	03-Aug 93	268	66	102	84.16	10.92
12 km	10-Aug 93	220	68	105	84.7	12.61
12 km	17-Aug 93	174	68	104	84.99	11.55
0 km	20-Jul 94	81	52	90	71.37	8
0 km	28-Jul 94	66	56	97	72.21	8.93
0 km	25-Aug 94	12	59	88	74.17	8.58
0 km	31-Aug 94	43	62	83	75.21	8.41
0 km	09-Sep 94	6	68	83	76.5	8
0 km	14-Sep 94	8	67	85	75.75	6.72
0 km	28-Sep 94	8	73	97	82.75	7.01
0 km	11-Oct 94	2	90	91	90.5	6.03
12 km	02-Jun 94	41	48	87	61.05	6.39
12 km	03-Jun 94	55	44	86	60.4	7.02
12 km	04-Jun 94	56	47	90	62.89	6.99
12 km	05-Jun 94	28	50	88	63.82	6.24
12 km	06-Jun 94	106	45	95	62.62	6.28
12 km	07-Jun 94	14	48	76	57.14	6.61
12 km	08-Jun 94	48	43	86	58.79	9.97
12 km	09-Jun 94	8	53	79	62.75	8.58
12 km	10-Jun 94	14	47	91	65.43	5.57
12 km	11-Jun 94	25	48	94	68.68	5.77
12 km	12-Jun 94	35	52	91	69.26	6.43
12 km	13-Jun 94	41	45	91	70.61	5.36
12 km	14-Jun 94	21	50	93	70.24	6.06
12 km	15-Jun 94	41	54	93	70.37	8.04
12 km	16-Jun 94	56	46	92	68.21	7.12
12 km	17-Jun 94	84	46	104	71.43	8.33
12 km	18-Jun 94	53	49	87	72.26	7.39
12 km	19-Jun 94	53	49	87	72.26	7.38
12 km	20-Jun 94	11	66	82	76.82	5.55
12 km	21-Jun 94	34	57	95	78.44	5.09
12 km	22-Jun 94	12	69	98	83.33	5.2
12 km	23-Jun 94	14	69	91	82.21	4.89
12 km	24-Jun 94	14	58	87	78.57	5.93
12 km	25-Jun 94	25	66	91	79.84	6.54
12 km	27-Jun 94	97	58	97	76.95	4.65
12 km	28-Jun 94	132	49	94	76.83	4.91
12 km	29-Jun 94	169	50	94	75.66	6.78
12 km	30-Jun 94	170	42	93	74.22	5.67

Appendix I (Continued).

Site	Date	Count	Fork Length (mm) of Chinook			Std. Deviation
			Minimum	Maximum	Mean	
12 km	01-Jul 94	126	45	96	76.29	8.41
12 km	02-Jul 94	81	46	94	73.84	5.27
12 km	03-Jul 94	69	53	93	73.46	6.42
12 km	07-Jul 94	98	49	90	74.9	8.33
12 km	08-Jul 94	76	59	93	73.72	3.32
12 km	09-Jul 94	170	54	93	75.99	4.42
12 km	10-Jul 94	126	47	96	74.56	8
12 km	11-Jul 94	114	55	87	73.13	6.72
12 km	12-Jul 94	116	55	90	75.49	0.71
12 km	13-Jul 94	123	59	92	74.74	7.45
12 km	14-Jul 94	195	59	96	74.93	6.39
12 km	15-Jul 94	203	57	93	75.17	7.62
12 km	16-Jul 94	180	58	96	75.23	6.99
12 km	17-Jul 94	168	58	94	74.83	6.24
12 km	18-Jul 94	206	57	92	74.47	6.28
12 km	19-Jul 94	102	61	92	75.96	6.61
12 km	21-Jul 94	60	61	89	76	5.57
12 km	22-Jul 94	65	63	92	75.43	5.77
12 km	26-Jul 94	100	59	93	75.95	6.43
12 km	28-Jul 94	80	64	88	75.9	5.36
12 km	29-Jul 94	93	61	89	75.92	6.06
12 km	30-Jul 94	124	56	97	75.24	7.12
12 km	03-Aug 94	104	51	90	75.63	6.78
12 km	09-Aug 94	43	66	89	75.44	8.17
12 km	10-Aug 94	72	61	91	76.86	5.55
12 km	11-Aug 94	57	66	90	77.26	5.09
12 km	12-Aug 94	46	67	87	76.07	5.2
12 km	15-Aug 94	30	64	85	76.23	4.89
12 km	24-Aug 94	12	64	87	77.25	5.44
12 km	26-Aug 94	31	69	89	77.81	4.65
12 km	27-Aug 94	27	70	86	77.7	4.91
12 km	30-Aug 94	29	67	91	78.62	6.82
12 km	11-Sep 94	15	66	92	81.67	6.42
12 km	20-Sep 94	4	82	89	84.5	3.32
12 km	27-Sep 94	9	81	95	88.33	7.38
12 km	13-Oct 94	32	75	102	89.75	5.89

Appendix J. Mean fork length (mm) and variance of chinook captured at two trap sites on the Middle Fork Smith River, California with t-test results. Weeks of July and August are consecutively numbered for (A) 1993 and (B) 1994. Mean difference in size per year is in bold.

Week	Site 0 km				Site 12 km				difference	var(pooled)	T-stat	df	Pvalue
	Date	n	x	var	Date	n	x	var					
A. 1993													
1	03-Jul	30	68.67	72.5904	29-Jun	22	74.73	62.41	-6.06	68.314632	-2.612077	50	0.005
2													
3	16-Jul	270	70.05	104.04	13-Jul	32	75.75	70.0569	-5.7	100.528413	-3.040772	300	0.001
4	30-Jul	61	77.05	68.3929	28-Jul	287	84.1	83.9056	-7.05	81.2155364	-5.54863	346	0.001
5	06-Aug	54	80.89	53.5824	03-Aug	268	84.16	119.2464	-3.27	108.3708	-2.105853	320	0.017
6	12-Aug	92	81.39	83.7225	10-Aug	220	84.7	159.0121	-3.31	136.910959	-2.278436	310	0.012
7	18-Aug	41	79.68	42.7716	17-Aug	174	84.99	133.4025	-5.31	116.382613	-2.835292	213	0.002
8									-5.116667				
B. 1994													
1	20-Jul	81	71.37	64	18-Jul	206	74.47	39.4384	-3.1	46.3328842	-3.47258	285	0.001
2	28-Jul	66	72.21	79.7449	26-Jul	100	75.95	41.3449	-3.74	56.5644122	-3.13558	164	0.001
3													
4													
5	25-Aug	12	74.17	73.6164	24-Aug	12	77.25	29.5936	-3.08	51.605	-1.05022	22	0.147
6	31-Aug	43	75.21	70.7281	30-Aug	29	78.62	46.5124	-3.41	61.04182	-1.816384	70	0.034
7									-3.3325				
8													

Appendix K. Correlation coefficients (r) for chinook and steelhead >1 counts between sampling occasions in the Middle Fork Smith River, California. Common samples were compared by habitat type.

Habitat	Chinook			Steelhead >1		
	'93.1/'93.2	'94.1/'94.2	'94.2/'94.3	'93.1/'93.2	'94.1/'94.2	'94.2/'94.3
Pool	--	-0.1284	0.7674	--	0.9182	0.9713
Pool Head	0.7771	0.4513	0.6160	0.8337	0.3752	0.7943
Pool Body	0.3957	-0.1843	0.5245	-0.4910	0.9753	-0.0598
Pool Tail	-0.4474	0.8928	0.7634	0.7240	0.8538	0.9865
In-pool Run	--	0.3377	0.5983	--	0.9508	0.9656
Rif. Margin	0.6256	0.6941	0.4990	0.8762	0.8379	0.8344
Rif. Mid.	--	0.6006	--	0.5937	0.9226	0.9705
Run Margin	0.1875	0.3621	--	0.1687	0.6000	--
Run Mid.	0.9610	0.3551	--	0.8549	0.2325	--
Average	0.4166	0.3757	0.6281	0.5086	0.7407	0.7804