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J. B. Scott, C. R. Steward, and Q. J. Stober

TECHNICAL COMPLETION REPORT
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Roy E. Nakatani

for

R. L. Burgner, Director
Fisheries Research Institute

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1.0 ABSTRACT

A three-year study was conducted to assess the impact of urban development upon the fisheries resources of Kelsey Creek, Bellevue, Washington. Streambed scour was significantly greater than in a control stream during periods of stormwater runoff, but the resultant mortality of coho salmon and cutthroat trout embryos was estimated to average less than 15 percent in the years 1978-1981. The composition of the streambed varied in response to hydrologic events, becoming significantly coarser during the winter months. Little difference existed in the percentage of fine sediment ($< .841$ mm) in substrate samples from Kelsey Creek and the control stream. Intragravel dissolved oxygen concentrations in the urban stream were typical of those in disturbed watersheds and were inadequate to meet salmonid embryo respiratory requirements. The mortality of coho salmon embryos in instream bioassays was significantly greater in Kelsey Creek than in the control, but no difference was found in the survival of rainbow trout embryos.

The biomass of juvenile salmonids supported by Kelsey Creek was generally greater than that in the control stream, ranging from 1.8 to 6.5 $g \cdot m^{-2}$. However, the majority of this biomass was age 0 and 1 cutthroat trout, while the control stream supported a diverse assemblage of salmonid species-age classes, and numerous nonsalmonids as well. Growth of salmonids in Kelsey Creek was excellent, and annual production was estimated as 6.6 and 7.4 $g \cdot m^{-2}$ in 1979 and 1980, respectively. Marking and outmigrant studies indicated that the unstable environment of Kelsey Creek did not result in the voluntary or forced displacement of the salmonids inhabiting the stream.

KEYWORDS: urbanization, streams, scour, substrate, embryos, salmonids, production, stability.

2.0 ACKNOWLEDGEMENTS

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No study of the length and intensity as that reported herein is possible without the diligent efforts of many people. Perhaps most important were the landowners who graciously allowed access to the study streams over an extended period of time. Numerous field assistants, including Bruce Johnson, Son Nguyen, Chinh Cao, and Bruce Ault, were frequently called upon to work long days, often on "holidays." Their conscientious work is much appreciated.

The professionals at the United States Geological Survey (USGS), the Washington State Department of Fisheries (WDF), METRO, and the City of Bellevue, of whom O. Hettick (USGS), B. Wiggins (USGS), D. Allen (WDF), K. Reidinger (WDF), A. Johnson (METRO), P. Bissonnette (City of Bellevue), and H. Brende (City of Bellevue) were foremost, provided data at timely intervals and assisted in the operation of many of the field projects. Special attention should go to P. O'Flaherty (City of Bellevue) for her unflagging efforts to coordinate projects and expedite public participation. Students at the Bellevue Christian, Interlake, and Bellevue High Schools were instrumental in site preparation and maintenance of the streamside incubation boxes. Members of the Eastside Steelheaders and Overlake Fly Fishing Club were helpful in this regard as well.

Finally, our colleagues in the Department of Civil Engineering, Dr. M. A. Perkins, J. Richey, and R. Pedersen, are to be thanked for their insights, encouragement, and patience.

3.0 INTRODUCTION

A significant outgrowth of the 1972 Federal Water Pollution Control Act Amendments (Public Law 92-500) has been a departure from a technological, standards-oriented approach to water pollution control to emphasizing the ecological integrity of aquatic systems (Squires 1975, Perkins et al. 1978). It has become increasingly evident that resource protection requires a basic awareness of the potentially adverse effects caused by pollution upon the biota of receiving waters, in addition to an understanding of the nature and sources of pollutants entering our waterways. Past efforts to improve water quality have been traditionally directed toward the assessment and technological solution of "point source" discharges, such as industrial and municipal effluents, under the assumption that loadings related to such discharges are the major contributor to water quality deterioration. While this undoubtedly has been and continues to be true in many cases, the abatement of these pollution sources has shown that non-point source (NPS) discharges must be controlled if water quality management goals are to be attained (Litwin and Donigian 1978).

In contrast to point source waste discharges, which are somewhat amenable to study and treatment, non-point pollution is a non-discrete, diffuse entity which is more difficult to assess because of its complex and variable nature. Sources of non-point pollution arising from human land use activities include agriculture, mining, silviculture, construction, and urban development (Manning et al. 1977), each contributing a variety of contaminants to receiving waters (McElroy 1975). Perhaps the most diverse collection of pollutants entering aquatic systems arises from urban areas, derived from sources such as waste petroleum products, lawn fertilizers and pesticides, atmospheric dust fallout, and land erosion.

Factors which determine the impact of NPS pollution on the aquatic environment include the rate at which contaminants are generated, the mechanisms by which the pollutants are transported across the land surface, and the assimilative capacity of the receiving waters. Porcella and Sorenson (1980) have summarized the relationship between these factors and watershed, hydrologic, and transport phenomena:

1. Watershed factors include percent impervious area, traffic and housing densities, industrial activity, and street sweeping activity.
2. Hydrologic variables include seasonal patterns of storm intensity and duration, annual precipitation, and the interval between storms.
3. Transport factors include the storm runoff collection and discharge system and the presence of separate or combined sewers. Transport processes and the hydrologic regime of the stream combine to determine the stream concentration for a given load of contaminant.

Numerous studies provide detailed descriptions of the effects that urbanization and stormwater runoff have on the physical and chemical conditions in streams. Documentation of these effects, however, have dealt primarily with the descriptive characterization of runoff itself, and until recently has given little consideration to the impacts of urban runoff on the stream biota, their interrelationships and habitat requirements. The analysis of the effects of urban development on stream ecosystems is a necessary step if we are to ultimately fulfill the objectives of P.L. 92-500, that is, "the protection and propagation of all forms of life that associate themselves with water" (Mackenthun 1975).

A three-year study was jointly initiated by the Fisheries Research Institute and Department of Civil Engineering, University of Washington, in 1979 with the intent of providing a more detailed and precise description of the ecological impacts of urbanization on a natural stream. The major objectives of the study, as outlined in the initial proposal, were to

1. Document the existing condition of an urban stream in terms of the spatial and temporal variation in the distribution and abundance of aquatic organisms;
2. Compare the condition of the urban stream with that of an undisturbed stream located within the same general area;
3. Evaluate the effects of urbanization using biotic indices and physical/chemical characteristics;
4. Evaluate the ecological influence of specific storm runoff events; and
5. Determine those factors limiting the survival and production of resident and anadromous salmonids in urban streams.

The major thrust of the fisheries investigation has been concerned with the assessment of salmonid habitat quality and structure, and the evaluation of salmonid population dynamics and streamflow requirements as they are affected by urban runoff. Throughout the study we have attempted to relate specific features of the salmonid life cycle to temporal and spatial changes in the urban stream environment. The reader is referred to an earlier progress report (Perkins et al. 1980) for additional information on the fisheries aspects of the investigation, particularly in regard to salmonid spawning and egg incubation studies. Other project-related material which has been published is summarized in Section 5.2.

4.0 LITERATURE REVIEW

Little research has been conducted with the specific intention of determining the effect of urban development upon fish populations in receiving waters. However, the attributes and effects of the various constituents of urban non-point source pollution have been the focus of numerous studies, and these will be reviewed below.

4.1 Urban Hydrology

An important characteristic of the conversion of rural lands to urban communities is the construction of residential, industrial, and commercial buildings and facilities, with the resultant increase in impervious area. Although urbanization may increase the volume of precipitation within a watershed, the primary hydrologic impact of development is a decrease in the infiltration of rainfall and, consequently, an increase in surface runoff and erosion (Guy 1972).

Under natural conditions a portion of the precipitation falling on an area is intercepted by vegetative cover. Interception becomes less important with increasing storm intensity and duration. As rainfall continues, infiltration occurs until the retention capacity of the soil is exceeded. The amount of infiltration is influenced by the type and depth of the soil mantle, the root development of vegetation present, and the degree that the ground is saturated by previous precipitation (Conley 1974). Soils covered with vegetation are quite permeable and extensive periods of rainfall are usually necessary to saturate the soil. When the amount of precipitation is greater than the storage capacity of the soil, or when rainfall intensity exceeds the infiltration rate, the excess water begins to fill surface depressions. Overland flow begins when the puddles, ditches, and other depressions become filled (Rendon-Herrero 1978).

Urbanization dramatically alters the rainfall-runoff relationship in a watershed. The reduction in vegetation minimizes the importance of interception storage and transpirational water loss. Infiltration characteristics are modified by the impervious area in the basin and the degree of compaction of exposed soil. Depression and channel storage are reduced through grading for more efficient drainage and the provision of man-made channels such as storm sewers which transport runoff to the stream (Conley 1974).

The regimen of stream flows in an urban watershed is governed primarily by runoff patterns. A principal factor is the rate at which runoff is transmitted across the land to the stream channel. The velocities of water transported through man-made channels is much greater than across areas covered with vegetation. An increase in the imperviousness of the watershed causes a decrease in the time interval between the center of a storm mass and the center of the resultant hydrograph mass (Anderson 1968, Ellis 1976). This effect is illustrated in Figure 1. The decrease in basin lag time in turn enhances the intensity and volume of runoff from the watershed. Several authors have noted that as runoff is concentrated in time, the peak rate of runoff (flood peak) increases (Cech and Assaf 1976, Gundlach 1976, Park 1977, Hossain et al. 1978). For example, under conditions of uncontrolled development in the Menomenee River watershed in Wisconsin, Walesh and Videkovich (1978) observed increases of up to 4.5 times in predicted 100-year flood stages. Hollis (1975) argues that smaller, frequent floods experience the greatest increase (up to 220 percent for the maximum monthly flood).

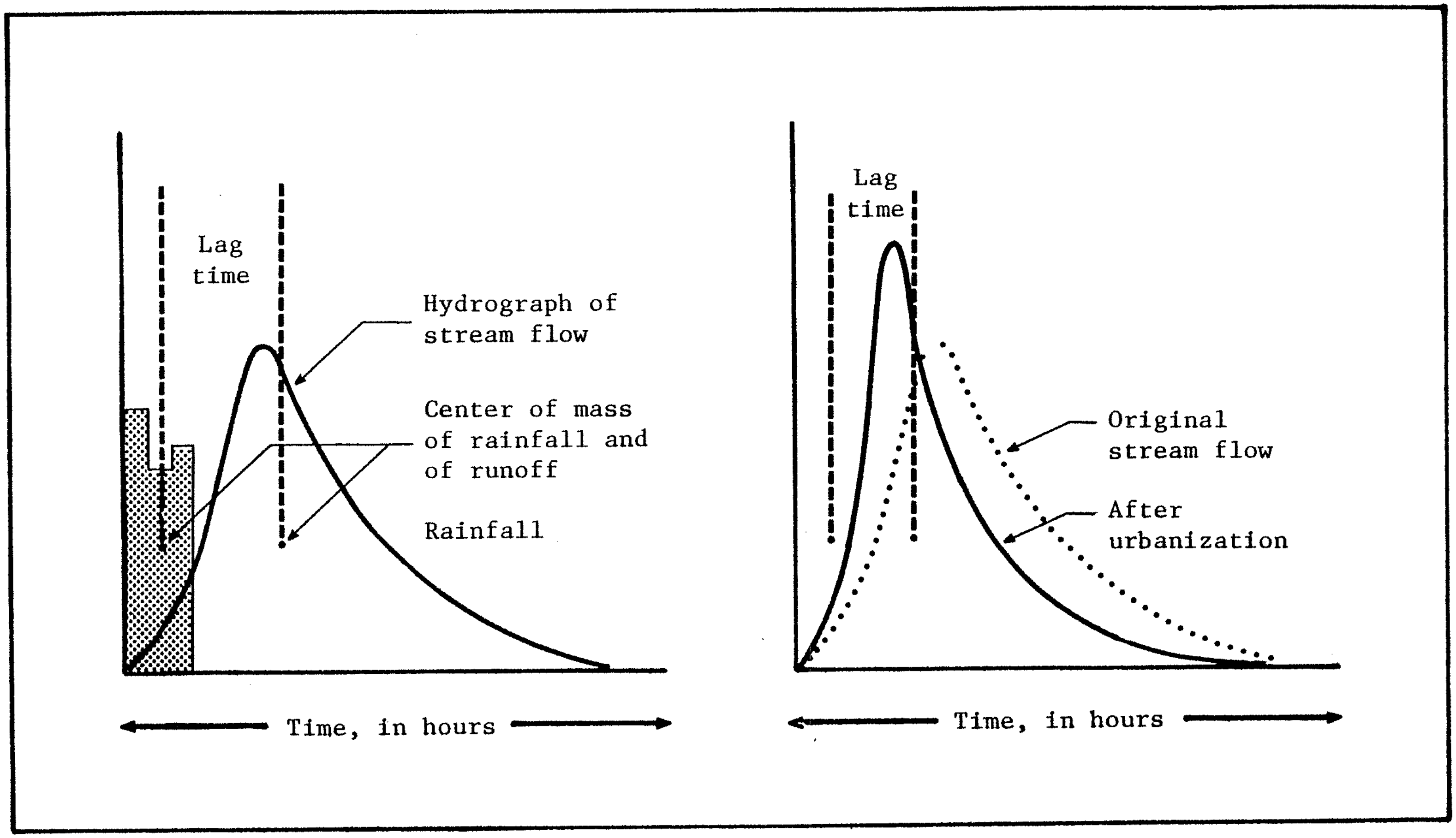

Figure 1. Hypothetical relationship between precipitation and runoff in an urban watershed. Redrawn from Metro (1976).

In addition to determining peak flows, the volume of water delivered as storm runoff affects low flow conditions in the urban stream. This results from the smaller volume of water available for recharge of natural groundwater, a lowering of the water table, and a reduction in the outflow of groundwater to augment low stream flows (McGriff 1972). Therefore, changes in runoff associated with urbanization have the effect of increasing flood peaks during storm events and decreasing low flows between storms.

4.2 Sediment Production, Transport, and Storage

It is generally recognized from field and laboratory studies that abnormally high levels of fine sediments have a deleterious effect on fish and their freshwater habitat. The general effects of inorganic sediment on the aquatic life of salmon and trout streams have been thoroughly reviewed in the literature (Cordone and Kelly 1961, Hollis et al. 1964, Everhart and Duchrow 1970, Phillips 1971, Koski 1972, Gibbons and Salo 1973, Meehan 1974, Mortensen et al. 1976, Iwamoto et al. 1978, Newbold 1980, Scullion and Milner 1980). A summary of potential effects of sedimentation on stream-living salmonids follows, adapted from Iwamoto et al. (1978):

1. Clogging and abrasion of gills;
2. Abrading or adhering to the egg chorion;
3. Increasing susceptibility to disease;
4. Behavioral modification;
5. Blocking emergence of alevins;
6. Reducing spawning habitat availability;
7. Affecting intragravel permeability, dissolved oxygen concentrations, and metabolic waste transport;

8. Altering water chemistry through the adsorption and/or absoprtion of ionic substances; and
9. Altering the structure and productivity of the food resources available to fish.

The problems of erosion and sedimentation are closely related to the hydrologic alterations of urban development. Changes in both discharge and sediment transport, related to construction practices during different phases of urbanization, have been summarized by Guy (1972), Leopold (1972), and Park (1977). As might be expected, sediment loading to urban streams has been found to vary according to construction practices and physical site characteristics. The most striking effects on production have been observed during periods of active construction, when sediment loads may be increased between five and tenfold (Guy and Ferguson 1962, Walling and Gregory 1970). Studies by Guy and Ferguson (1962) in the Washington, D.C. area have shown that an average sediment yield of 25,000 tons per square mile resulted from construction activities during a 19-year period. Wohlman and Schick (1967) present a summary of sediment yield data for rural and urban streams in the eastern United States which shows that sediment yield from urbanizing areas ranged from 1,000 to 100,000 tons per square mile per year. Figure 2 summarizes sediment yield data for areas of different sizes and land masses in the United States and gives some idea of the influence of human activity on the relative stability of land under natural cover, agriculture, and urban development.

A considerable amount of effort has been directed toward understanding the relation between changes in bed composition and the degree and quality of land use activities within the watershed (Brown 1971, 1976; Brown and Krygier 1971; Moring 1975a, 1975b; Moring and Lantz 1974, 1975; Cederholm et

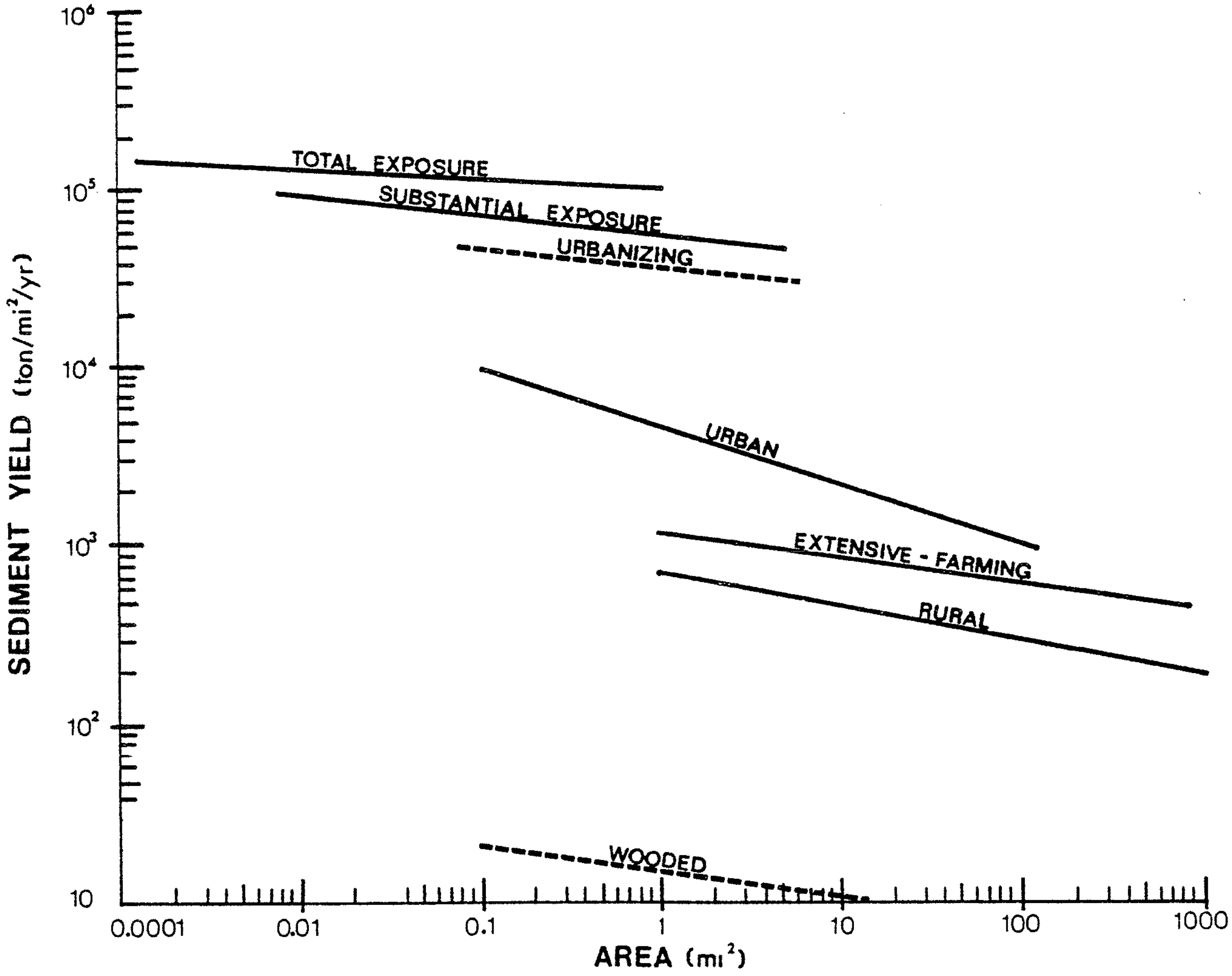

Figure 2. Sediment yield for areas of different sizes for a range of land use activities in the United States (redrawn from Manning et al. 1977).

al. 1977). As regards the impact of watershed disturbances on salmonid resources, however, the focus has been on the effects of forest harvesting activities on the ecology of coastal streams in the northwestern United States. Studies specifically addressing the response of urban streams, in terms of the quality and availability of spawning habitat, to changes in hydrologic and sediment dynamics associated with urbanization have not been reported in the literature.

Investigations concerned with the impacts of forest harvesting practices on streambed composition have been reviewed extensively elsewhere (Gibbons and Salo 1973, Adams 1980) and need not be restated here in detail. The general conclusions are that tree-felling, road construction and maintenance, yarding, and slash burning cause increased water yield, runoff rate, erosion, and stream sedimentation (Scullion and Milner 1980). The construction of logging roads, in particular, has been judged the major cause of increased sediment levels observed following logging activities (Lantz 1971; Swanston 1971; Swanson and Dryness 1975). An increase in mass wasting, or landslides, resulting from slope destabilization is another prominent source of sediment (Megahan and Kidd 1972; Cederholm and Lestelle 1974; Cederholm et al. 1975, 1977, 1978; Cederholm and Salo 1979). It is important to point out that the initially high rates of erosion and sedimentation caused by deforestation tend to approach pre-treatment levels within a decade following logging and burning. The eventual recovery of streams in disturbed watersheds has been attributed to revegetation of denuded areas and a decrease over time in logging-related slope failures (Megahan 1974, Brown 1971, Beschta 1978).

In a manner similar to that for forest harvesting activities, urbanization increases sediment production by disturbing the land surface so that it is more susceptible to erosion and by increasing the variability of water runoff patterns. Most of the sediment (> 50 percent) transported during the year is during periods of high flow; this has been documented for both natural drainages (McPherson 1971, Paustian and Beschta 1979) and urban watersheds (McGriff 1972). A single storm event in Sydney, Australia, lasting less than an hour resulted in runoff from 13 mm of precipitation that carried 1150 kg suspended sediment from a 131 km^2 area (Cordery 1977). Colston (1974), in a report on runoff quality in the Third Ford Creek watershed within the City of Durham, North Carolina, presented data on total suspended solids concentrations during 36 storms. Sediment transported during peak flows ranged from 83 to 3913 $mg \cdot l^{-1}$. The suspended sediment loads reported for urban streams contrast sharply with the relative magnitudes reported for rivers draining undisturbed watersheds. Cooper (1965) summarizes suspended sediment data collected from 46 western rivers during the period 1906-1913. The most frequently observed concentrations were in the range of 0-10 $mg \cdot l^{-1}$, and 90 percent of all the concentrations recorded were less than 150 $mg \cdot l^{-1}$.

There is some indication that sediment yield from urban areas may decrease as some semblance of soil stability is regained. Runoff in Tucson, Arizona, contained less suspended sediment as residential areas became more developed (Mischi and Dhasmadhikari 1971). In the study by Guy and Fergusen (1962), mentioned earlier, the storm runoff in streams draining urban and suburban areas appeared to return to normal sediment yields following stabilization. In urban areas, however, the diminished contribution of soil erosion may be replaced by solids generated as a product of other human activity. For example, it has been estimated that only 0.25 mm of concrete pavement surface erosion annually would produce a potential sediment yield of 2 tons/

acre/year or more of pavement material (Manning et al. 1977).

From the data presented here, it is clear that the substantial sediment yield from urban watersheds is transported into and through the channel system as a discrete mass or wave. The interaction of hydrologic factors and sediment loads may cause changes in the channel morphology of the urban stream. These, in turn, may alter the hydraulic properties of the channel, cause bank erosion, and affect the bed composition. A prerequisite for understanding the effects of urbanization on salmonid spawning habitat is a knowledge of the basic hydraulic and geomorphic processes which determine stream bed composition.

The dynamic interaction between sediment concentrations, hydraulic variables, and bed composition involve mechanisms which are not yet fully understood. Investigations of sediment transport and deposition have often been confined to laboratory flumes where the variables of depth, width, bed composition, and sediment characteristics are necessarily controlled. Unfortunately, field conditions are generally more heterogeneous and, therefore, much more complex.

The behavior of sediment in a stream is determined by the relative strength of flow-generated forces and stabilizing forces. In the case of suspended sediment, the lift and drag produced by turbulent velocity fluctuations must overcome the gravitational forces acting on the immersed particle. In motionless water all sediment eventually settles to the bottom; however, the rate of settling is strongly dependent on particle shape, volume, and specific gravity. Lane (1938) showed that for particles less than 0.1 mm in size settling velocities varied as the square of the diameter. The settling velocities of particles greater than 0.3 mm varied as the square root of the diameter, with a transition zone from 0.1 to 0.3 mm. Thus, sand settles quickly, whereas clay particles settle very slowly. In flowing water where turbulence is present to some degree, there is usually sufficient energy to keep particles less than 0.1 mm in suspension (Cooper 1965). The effect of turbulence is to superimpose a randomly fluctuating vertical component on the settling velocity of a particle. Einstein (1968), in a series of flume experiments, determined that the sediment size, depth of water, and local sediment concentration were the most significant factors affecting rate of deposition.

If water turbulence near the bed is great enough, particles lying on the substrate surface may be projected upward into the flow. In an earlier study, Einstein (1950) attributes bed erosion properties to the influence of sediment size and turbulence. The stabilizing force of particle weight may be augmented in surface sediments by the attraction between colloidal particles. The cohesiveness of clay and organic particles makes the bed far more resistant to scour than would be the case if submerged weight were the only factor (Leytham et al. 1979). To date, however, the effects of particle shape, its position in the bed, and its interaction with other particles are not satisfactorily accounted for by theories of bed erosion (Scullion and Milner 1980).

In-channel sediment transport consists of a multitude of individual particles, each characterized by its own trajectory. The movement may be confined to the water column, essentially free of the stream bed, as is the case of suspended sediment. The dominant modes of travel, however, involve some form of contact with the substrate; larger particles generally move with a rolling, sliding, or saltatory motion (Grant 1974). Some authors (Graf 1971, Shen 1971) have indicated that in many situations the sediment load carried in this fashion may comprise 90-95 percent of the total load transported in the stream.

In addition to the factors discussed above, the fluvial behavior of sediment is influenced by the composition and form of the bed. A sediment particle normally carried in suspension may be deposited if it enters the laminar sublayer near the stream bed surface where its settling velocity will be sufficient to allow deposition. The thickness of this "dead water" zone, which varies according to velocity, viscosity, and turbulence, is a few millimeters. This section of flow is characterized by minimal velocities resulting from the frictional forces of the stream bed. Fine particles will remain as surface sediment only as long as they are shielded from the influence of turbulence by irregularities in the bed surface (Einstein 1950).

In streams containing a mixture of bed materials of different sizes, high flow conditions may result in an armor layer of coarser material which has become imbricated and fixed in position, while residual surface fines are winnowed away. Armoring affects the availability of in-channel sediment since it protects the underlying material from scour (Paustian and Beschta 1979, Griffiths 1980). According to Garde et al. (1977), the major coarsening of the surface layer in a degrading system occurs in a relatively short time period, after which the coarsening process is extremely slow. Urban streams are prone to armoring if their beds contain a sufficient amount of coarse material to provide resistant armor coating. After the drainage area has stabilized the new channel regime may be expected to have a coarser median particle size than prior to urbanization (Guy 1972).

The formation of an armor layer has the additional effect of inhibiting the intrusion of fine sediments into the stream bed. Although sediment intrusion into armored beds is not well understood, the general process of intrusion has been described from several laboratory studies. Sediment deposition into an artificial gravel bed was investigated by Einstein (1968), who concluded that the coarsest particles settled out first, filling the gravel interstices from the bottom upwards. Experimental flows kept the gravel surface free of sediment. More recently, Beschta and Jackson (1978) studied the amount of intrusion of two different grades of sand into gravel beds under varying conditions of discharge, depth, velocity, flume slope, and rate of sediment transport. Several factors were found to influence intrusion, including the Froude number of the flow, the rate of sediment transport, and sediment particle size. At low Froude numbers ($Fr < 0.9$) the 0.5 mm sand established an impermeable layer near the gravel surface which prevented further sediment intrusion. At higher Froude numbers ($Fr > 0.9$) the disturbance of the surface gravels by the flow caused the sand barrier to form deeper in the test substrates. The authors noted that a finer grade of sand, averaging 0.2 mm in size, filled the test gravels from the bottom up without forming an impermeable layer. It was concluded that the intrusion process may be selective toward smaller sediment sizes.

The shape of the bed materials may also have an appreciable effect on sediment deposition in spawning gravels. Meehan and Swanston (1977) demonstrated that round gravels trapped silt more effectively than angular gravels at low flows ($< 0.2\ m^3/s$). This relationship is reversed at higher flows ($> 0.4\ m^3/s$) and higher intrusion rates were observed for the angular gravels.

Cooper (1965) observed a relationship between intragravel flow velocities, determined by bed permeability measurements, and sediment deposition. Sedimentation occurs within the gravel even if velocities are too high to permit deposition on the bed surface. The lower intragravel water velocities associated with substrates of lower permeabilities allows sediment particles time to settle out of the flow. Sediment deposited in coarse gravels may also be substantial in spite of higher intragravel flow rates because it allows a larger volume of sediment-laden water to flow through them (Adams 1980). The filling of intragravel voids is a self-augmenting process if gravel permeability decreases and a supply of fine sediments exists.

The general pattern of sediment transport in both urban and forested watersheds is one of the large sediment loads being delivered to the stream during storms to be flushed en masse downstream by peak flows. If the supply of sediment is not equal to the carrying capacity of the flow, then the stream will either erode or deposit material (Bovee and Milhous 1978). Sediment may be accumulated and stored within the stream channel during low flow periods as a result of greater intervals between flood events, incomplete flushing of the sediment load by a previous storm, in-channel production of sediment such as bank erosion, and greater sediment input from human activities. Surface deposition and the intrusion of fine material into gravel interstices are the primary processes affecting bed composition during this period.

Another important consequence of high flows is the release of fine sediments from stream bed gravels. During initial peak flows, the sediment deposited on the surface of the stream bed is readily entrained in the water column. As the tractive forces of the flow increase, the armor layer is disturbed and the bed material set in motion. The disruption of the surface gravels and the subsequent increase in intragravel flow rates normally releases large amounts of fine sediments. If flows are great enough, the initial flushing out of sand and silt particles is followed by bedload movement of gravel and cobbles. Bagnold (1977) reported a rapid change in the median particle size of transported materials during a flood which he attributed to this process. In a study of the Silk Stream catchment of North London, England, Ellis (1976) noted rapid and efficient flushing of sediments during the rising limb of the storm hydrograph. Peak sediment loads and stream discharge were nearly coincident 20 to 30 minutes after the onset of precipitation. The bulk composition of the sediments was dominated by inorganic fractions (45-70 percent by weight) and were lognormally distributed in size. Less than 4 percent of the sediments were carried in suspension. The remainder, generally greater than 0.1 mm in size, was transported by saltatory movement.

After the hydrograph peak, the bed materials stabilize and inhibit the release of additional sediment. The deposition and intrusion of fine materials into the stream bed recommence during the falling limb of the hydrograph. An important feature of bed instability is that as a flood passes a given point the bed may be scoured as the flow rises and then refilled as it recedes (Paustian and Beschta 1978, Adams 1980).

Several authors have suggested the importance of the flushing process in determining the natural quality or composition of gravel substrates in temperate streams (McPherson 1971, Beschta 1978, Beschta and Jackson 1978, Adams and Beschta 1980). Others have noted a decrease in fine sediments in salmonid spawning areas during fall and winter months which they have attributed to the flushing effects of high flows and redd digging activities of spawning fish. For example, fall freshets dislodged and cleaned out the fine sediment which had been experimentally introduced into spawning riffles in Maybeso Creek, Alaska, reducing them to pretreatment levels (Shapley and Bishop 1965).

In two other Alaskan streams, McNeil and Ahnell (1964) and Helmers (1966) observed a pronounced reduction of fines in spawning gravels after a series of flood events. Similarly, high flows during the winter months scoured away heavy silt accumulations from the previous summer in Ellerslie Brook, Prince Edward Island (Saunders and Smith 1965).

In a time trend analysis of the bed composition in salmon and steelhead spawning areas of the South Fork Salmon River, Idaho, Platts and Megahan (1975) noted a marked decrease in the amount of fines in the riverbed after sediment loading from the watershed was curtailed. The recovery of an impacted stream, as it was pointed out, will occur only if sediment loads are reduced relative to the stream's capacity to flush fines from the system.

The massive amounts of sediment produced by two landslides in Stequaleho Creek, Washington, had a significant but short-term effect on the stream bed composition. The percentage of fines in the spawning gravels below the landslides returned to background levels after two winters (Cederholm and Salo 1979).

It is apparent that peak flows which mobilize the gravel bed and flush out fines are responsible in part for the temporal variability characteristic of stream beds. The flushing of intragravel fines is probably restricted to only part of the channel during peak flows since floods capable of scouring the bed over the entire length of the stream are rare (Adams 1980). The removal of fines by this process and the influence it has on the local bed composition will vary within and among streams in proportion to the energy available to entrain the sediment and the stabilizing forces present. The available energy over a given unit area of stream is governed by the discharge rate and hydraulic gradient, while the stabilizing forces are determined by the geomorphological properties (e.g., bed cohesiveness, sinuosity) of the channel. For example, the velocity and depth of flow near the edge of a stream are usually less than the velocity and depth near the center. It is logical to assume then that fine materials are deposited near the banks while relatively coarse material is found near the thalweg. Longitudinal changes in bed composition are introduced by changes in water velocity and channel pattern, as evidenced by the presence of riffles and pools in most salmonid streams. On a larger scale, there is generally a reduction in particle size in the downstream direction caused by a decrease in channel slope and bed shearing stress (Scullion and Milner 1980).

In summary, a number of complex and interrelated factors contribute to the production, transport, and storage of sediment. Bed composition may be expected to vary markedly in time and space in response to the local regimen of sediment load, flow, gradient, and related hydraulic parameters. On a watershed basis, the composition of stream gravels is influenced by the hydrological and geological characteristics of the drainage and its history of land use.

4.3 Scouring and Bed Stability

Streambed scour is one means by which fine sediments may be removed from the substrate. To a certain degree this is desirable; however, excessive scour may wash developing salmonid embryos from the streambed. It has also been noted (Adams 1979) that an increase in the depth of scour does not necessarily increase the effectiveness of the flushing action.

Two prerequisites are necessary before scouring can occur: 1) there must be particles available for transport, and 2) the flow must be capable of moving the particles. As Klingeman (1981) noted, the first condition will always be met in salmonid spawning areas. Defining more precisely the conditions under which scour will occur is exceedingly more difficult. In general, the threshold condition for particle motion, or critical shear stress, will depend upon the bottom width of the channel, stream depth, particle size distribution, particle density, the density and kinematic viscosity of the water, the shear velocity, and the acceleration due to gravity (Simons and Sentürk 1977). Relations derived for determination of critical shear stress are either very approximate (e.g., the Hjelstrom-Sunnborg diagram) or are dependent upon conditions unlikely to be met in the field (e.g., Shield's equation) (Hey 1976, Hall and See 1981).

Early observations of the impact of floods and scouring on salmonid embryos were largely descriptive. Numerous investigators (Wickett 1959, Davidson and Hutchinson 1974, Wittler 1952) have reported the occurrence of extensive deposits of eggs along the streambank during and after flooding. Neave (1953) speculated that the small 1944 chum salmon run to Vancouver Island, British Columbia, could be attributed to severe flooding during the fall of 1940. Streambed scour in Sagehen Creek, California, resulted in the displacement of nearly all salmonid eggs deposited prior to the flood (Needham and Jones 1959).

Perhaps the earliest quantitative estimates of flood-induced egg loss were made by Hobbs (1937), who sampled natural redds with a shovel and catch-net. He reported that a single flood completely washed away 27 percent of the redds under study; 23 percent were scoured to a depth of 7.6 cm, and one was scoured to a depth of 30 cm. Gangmark and Bakkala (1960) planted plastic enclosures containing chinook eggs in Mill Creek, California, and stated that floods in 1955 and 1958 caused losses of 100 and 98.2 percent, respectively, of the eggs. The pioneering research of McNeil (1962, 1966), who determined point estimates and confidence intervals for the mortality of embryos in southeastern Alaska streams, firmly established the critical role streambed scour plays in regulating embryo survival. In one stream area, egg loss to scour averaged 49 percent over a three-year period, and in several instances mortality was found to exceed 90 percent. Stober et al. (1978) applied similar techniques in the Cedar River, Washington, and estimated that embryo densities were reduced by 74 percent after a single large flood.

For a single watershed, a plot of the smolts (fry) produced per spawner versus peak discharge is a useful technique by which the critical discharge necessary to produce embryo mortality can be determined. For comparisons among watersheds, each of these quantities must be scaled appropriately. One simple approach is to express smolts per spawner for each year class as a percentage of the maximum value observed in that watershed, and peak discharge in terms of its annual recurrence interval. Data from two small watersheds (Deer and Flynn Creeks, Au 1972) and three larger ones (Minter Creek, Smoker 1955; Waddell Creek, Shapolov and Taft 1954; Cedar River, Stober and Hamalainen 1980), scaled and plotted by this procedure, exhibit contrasting results (Figs. 3 and 4, respectively, Table 1). Floods with recurrence intervals of up to 16 years had little or no effect upon the survival of embryos in the smaller watersheds. In the large streams, where peak discharges were 1 to 2 orders of magnitude above those of the small streams, the survival of embryos was sharply reduced by floods with recurrence intervals as small as 5 years.

Realistically, it must be noted that our ability to draw a general conclusion from the analysis is limited by the paucity of accurate data. While it appears that floods may inflict significant mortality upon embryos incubating in large streams, the relationship between streambed scour and flood recurrence interval/stream discharge in small streams is far less certain.

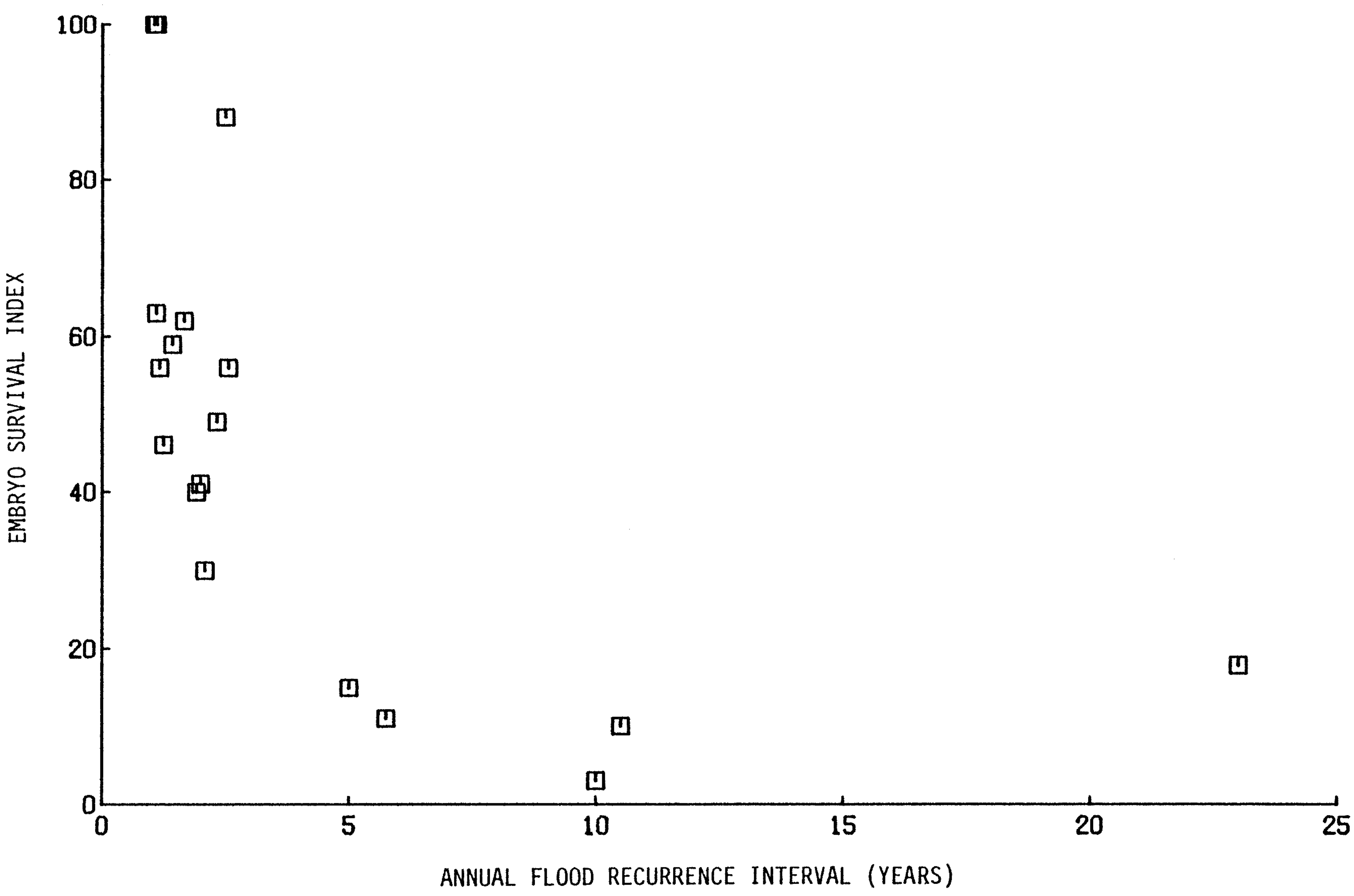

Figure 3. Impact of floods of various recurrence intervals upon the survival of salmonid embryos incubating in large streams (Minter Creek, Cedar River, and Waddell Creek). See Table 1 for sources.

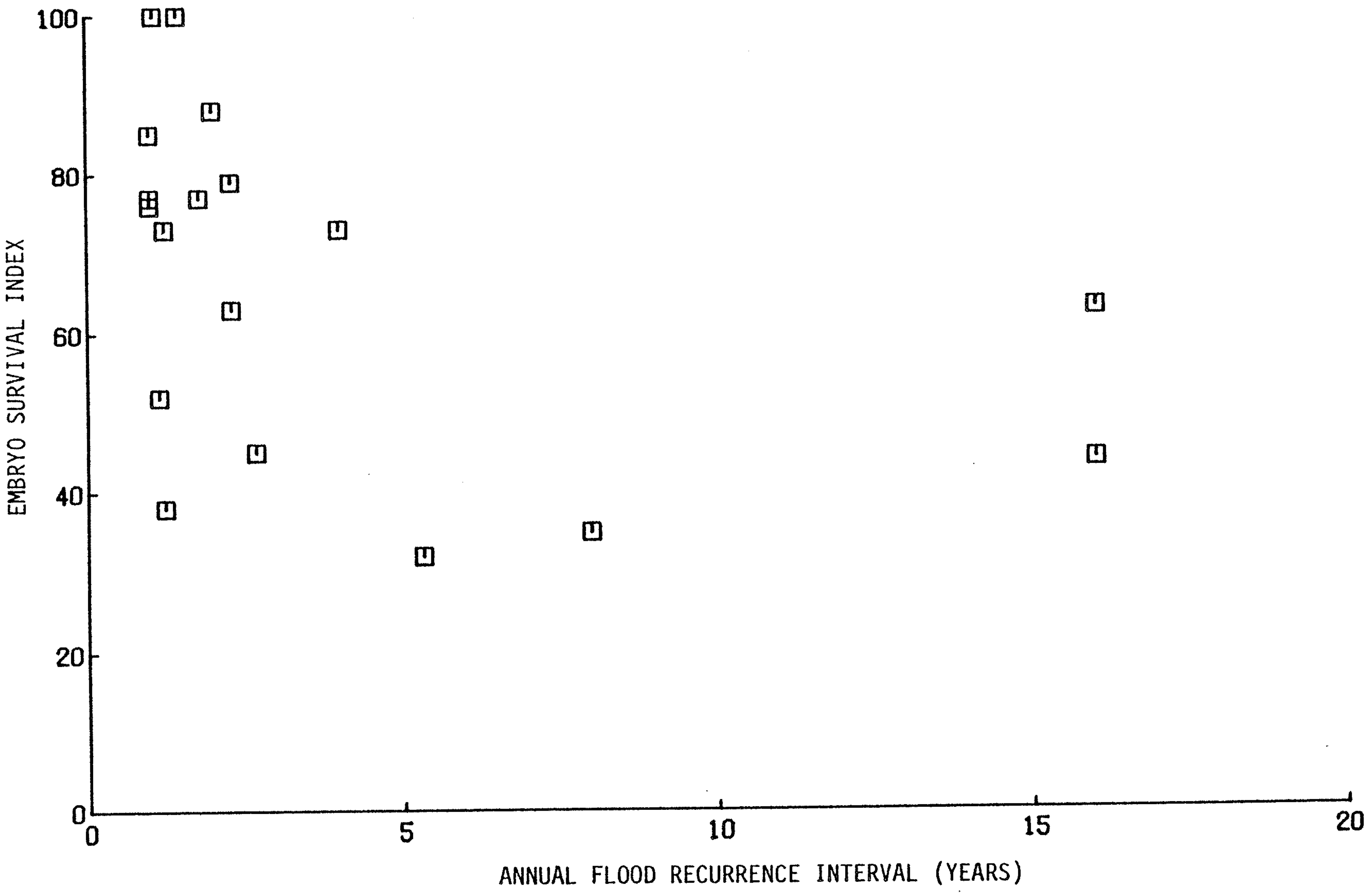

Figure 4. Impact of floods of various recurrence intervals upon the survival of salmonid embryos incubating in small streams (Deer Creek and Flynn Creek). See Table 1 for sources.

Table 1. Sources and data relevant to determining the relationship between flood recurrence interval, peak discharge, and embryo mortality.

Stream (State)	Water Data		Fisheries Data				
	Record	Q_{max} (m^3/s)	Species	Spawner index	Smolt/fry index	Period	Source
Small streams							
Flynn Creek (OR)	1959-1973	3.9[2]	Coho	Females	Migrant fry	1960-1969	Au (1972)
Deer Creek (OR)	1959-1973	5.7[2]	Coho	Females	Migrant fry	1961-1969	Au (1972)
Large streams							
Minter Creek (WA)	1947-1954[1]	28.3[2]	Chum	Females	Migrant fry	1948-1954	Smoker (1955)
Cedar River (WA)	1961-1980	249.0[2]	Sockeye	Escapement	Migrant fry	1965-1980	Stober and Hamalainen (1980)
Waddell Creek (CA)	1933-1941	182.8[3]	Coho	Females	Smolts	1933-1941	Shapolov and Taft (1954)

[1] Flood recurrence intervals computed from U.S.G.S. data for Huge Creek, the principal tributary to Minter Creek.

[2] U.S.G.S. data.

[3] Data from Shapolov and Taft (1954)

[4] Data from 1963 deleted due to the limited number of spawning females (2).

Direct measurements of streambed scour, both by geologists and fisheries biologists, have documented scouring of over 30 m in larger rivers (Leopold, Wolman, and Miller 1964), but for salmonid spawning streams, the potential depth of scour is considerably less. McNeil (1962) attempted to measure scour using plastic balls embedded in the streambed, but the results were inconclusive. The same technique was applied in the Alsea River watershed (Moring 1975). Yearly mean scour depths ranged from 1 to 18 cm, and mean yearly deposition ranged from 6 to 21 cm.

4.4 Intragravel Water Quality

A major portion of salmonid embryo research has sought to determine the supply rate and concentration of dissolved oxygen needed to meet embryonic respiratory demands. In general, oxygen consumption will vary with embryonic developmental stage and water temperature. Lindroth (1942) found that Atlantic salmon (*Salmo salar*) eggs near hatching consumed oxygen at a rate of 29 cc/kg/hr when incubated at 17°C, but only 16 cc/kg/hr oxygen were required when the temperature was reduced to 5°C. Oxygen consumption per egg also increases with developmental stage (Alderdice et al. 1958) up to the time of hatching. After hatching, the initiation of active respiration across the gill membranes acts to reduce the required level of dissolved oxygen. For example, at 10°C the development of Atlantic salmon eggs near hatching is retarded at oxygen concentrations of 7.5 mg/l or less. After hatching, concentrations may be reduced to 4.5 mg/l before alevin metabolism is affected (Hays et al. 1951).

Significant mortality is not induced until dissolved oxygen concentrations drop well below these values. In a two-way factorial experiment, Alderdice et al. (1958) exposed chum salmon eggs at various stages of development to reduced levels of dissolved oxygen. Median lethal dissolved oxygen concentrations ranged from 0.4 mg/l at 121.2 degree-days to approximately 1.2 mg/l at 452.4 degree-days (Alderdice et al. 1958). Similarly, Silver et al. (1963) demonstrated that at 9-10°C steelhead trout and chinook salmon egg survival was virtually unaffected by continuous exposure to dissolved oxygen concentrations as low as 2.6 mg/l. Both species experienced complete egg mortality when the dissolved oxygen level was reduced to 1.6 mg/l. Brannon (1965) reared sockeye salmon embryos from fertilization to yolk absorption in 8°C water under constant dissolved oxygen concentrations of 3.0, 6.0, and 11.5 mg/l. Mortality in each of the experimental lots was less than 6.0 percent.

In-situ experiments have often employed standpipes (Wickett 1954) to monitor intragravel dissolved oxygen levels (Ringler and Hall 1975, McNeil 1966, Woods 1980, Turnpenny and Williams 1980, Phillips and Campbell 1962, Coble 1961). An observation common to all these investigations was the extensive spatial and temporal variability of the results. Under these conditions, relating concentrations of dissolved oxygen determined in-situ to constant condition laboratory experiments may be misleading. Field studies (Coble 1961, Phillips and Campbell 1962, Turnpenny and Williams 1980) show that when dissolved oxygen levels are determined by the standpipe technique, concentrations below 8 mg/l may be expected to significantly reduce embryo survival (Fig. 5).

Surface stream water is the primary source of intragravel water (Sheridan 1962). For this reason, Vaux (1962) stated that the concentration of

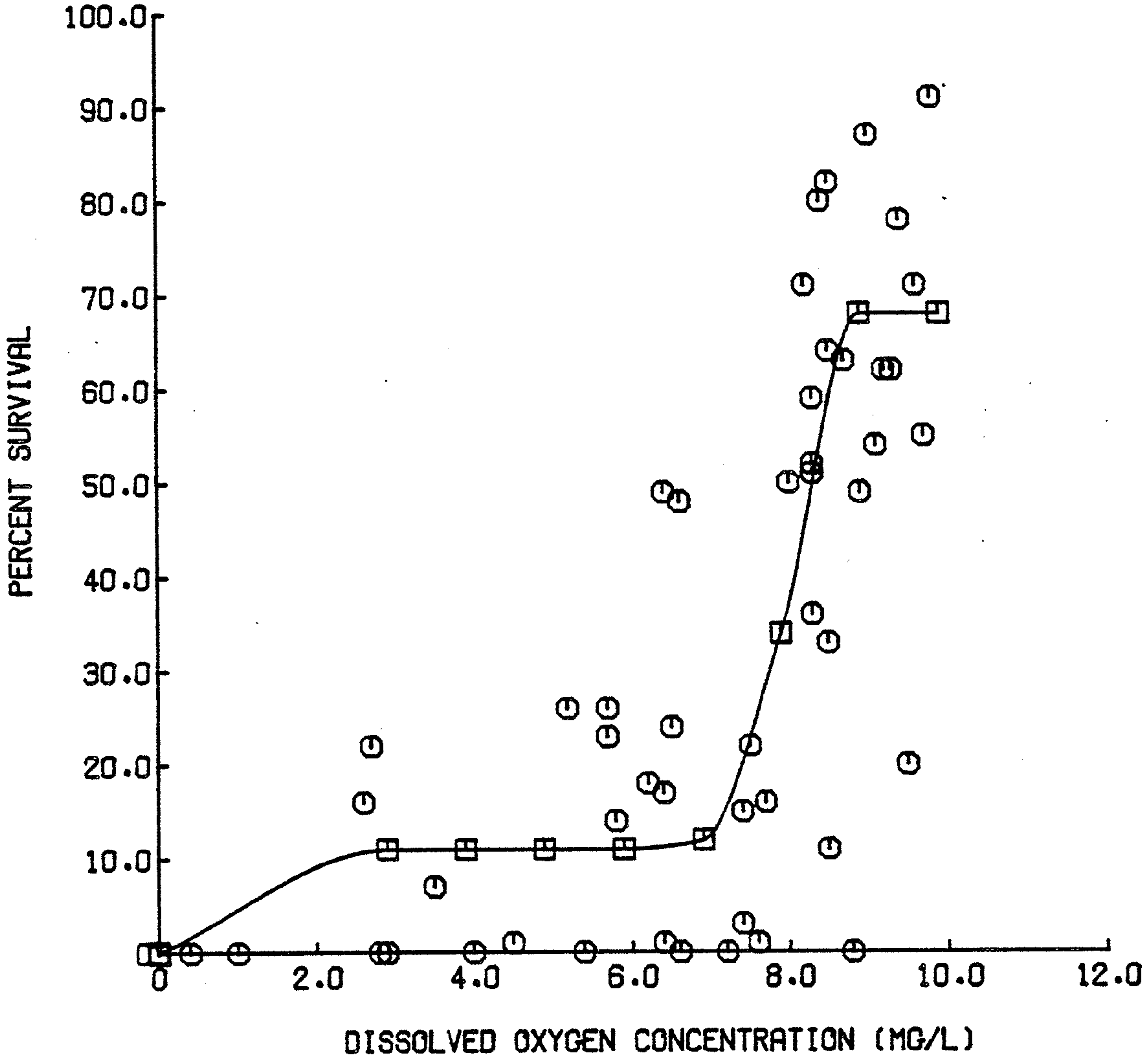

Figure 5. The relationship between intragravel dissolved oxygen concentrations and the survival of salmonid embryos. Line fit by eye (data from Coble 1961, Phillips and Campbell 1962, and Turnpenny and Williams 1980).

dissolved oxygen within the interstices is dependent upon 1) the diffusion of atmospheric oxygen into the surface water, 2) mixing of this oxygen throughout the water column, and 3) the exchange of surface and intragravel water. To these may be added the rate at which oxygen is produced and consumed in the surface water. In most streams, the penetration of surface water into the streambed is likely to be the limiting function. Factors important in regulating the exchange of water between the stream and gravel bed are stream surface profile, irregularity of streambed surface, gravel bed depth, and gravel permeability (Vaux 1962). The latter of these is particularly important since gravel permeability may be readily altered by watershed modification.

A medium is termed permeable if a fluid may pass through it, while the porosity of a material is the ratio of the volume of voids to the total volume of solids plus voids. The apparent velocity, or volume of liquid flowing per unit time through a unit area normal to the direction of flow, may be related to permeability by Darcy's Law:

$$V = ks$$

where V is the apparent velocity, k is the permeability coefficient, and s is the head loss per unit length in the direction of flow (hydraulic gradient). Thus, even in a highly permeable medium, the apparent velocity of the fluid will be zero in the absence of a hydraulic gradient.

The permeability coefficient is a function of the fluid viscosity and density, particle size, shape, and roughness, and the porosity of the medium. For a given fluid, Pollard (1955) determined that gravel shape is of negligible importance in determining gravel permeability, while McNeil and Ahnell (1964) experimentally demonstrated that a threefold increase in the percentage of gravel less than 0.833 mm reduced the coefficient of permeability by a factor of 10. Cooper (1965) showed that the addition of small particles (0.074 mm) had a greater effect upon permeability than did the addition of an equal weight of large particles (0.097-0.390 mm).

It may be concluded from these studies that the permeability of gravel is reduced by the addition of fine sediment which clogs the gravel interstices and restricts water flow. Impermeable gravels will be deficient in dissolved oxgyen due to the limited influx of surface water.

Direct sediment-induced reductions in fry fitness have also been documented. Several studies (White 1942, Koski 1966, Phillips et al. 1975) have suggested that alevins, which depend upon the interstices as an emergence route, become entombed when fine sediments accumulate. Phillips et al. (1975) found that when the volume of 1-3 mm sand was increased from 20 to 30 percent of the incubation medium, successful emergence declined from 65 to 40 percent. Partially clogged interstices also act as a selective force against large fry (Bjornn 1968, Koski 1975, Phillips et al. 1975), whose greater cross-sectional area increases the probability that emergence will be blocked.

Research conducted by Koski (1966), Hall and Lantz (1969), and Tagart (1976) suggests that the cumulative effect of these sediment-related factors on the survival of embryos from egg deposition to emergence can be quite significant. Koski (1972, quoted in Cederholm et al. 1978) stated "a 1% increase in sediment less than 0.833 mm resulted in a 45 percent decrease in survival to emergence."

Of all the toxicants observed in urban stormwater runoff, lead is the most prevalent. When dissolved in water it is readily available to fish for uptake. Hodson et al. (1978) found a strong correlation between ambient and tissue concentrations of lead in rainbow trout. Fish exposed to detrimental lead concentrations may develop hematological abnormalities (Hodson et al. 1978, 1980), blackened skin in the caudal area (Hodson et al. 1980, Davies et al. 1976, Holcombe et al. 1976), lordosis and scoliosis (Hodson et al. 1980, Davies et al. 1976, Holcombe et al. 1976), or muscle spasms (Davies et al. 1976). High concentrations are acutely toxic. Generally, salmonids are most sensitive to lead in soft water (EPA 1976) when exposed as alevins (Holcombe et al. 1976, Davies et al. 1976, Davies and Everhart 1973). Studies by Sauter et al. (1976) indicate that the eggs themselves may be somewhat more resistant.

A 96-hr TL50 (hardness 17-25 mg/1 $CaCO^3$) of 800 μg/1 lead was determined by Chapman (cited in U.S. EPA 1976) for coho four to five weeks after hatching; at an age of six to seven weeks this was reduced to 520 μg/1. Sauter et al. (1976) found that rainbow trout eggs exposed to a lead concentration of 672 μg/1 (hardness 45 mg/1 $CaCO_3$) had reduced hatchability, while the survival of alevins in a 30-day period after hatching was significantly reduced at lead concentrations of greater than 250 μg/1. Alevins which survived at these lead levels were smaller in length than controls. Scoliosis developed in 28 and 12 percent of the alevins reared at lead concentrations of 672 and 442 μg/1, respectively.

Temperature of intragravel water interacts with many factors in the spawning and incubation environment to influence the development, growth, and survival of salmonid eggs and alevins. High mortality has been observed in embryos subjected to low (< 4°C) temperatures immediately after fertilization (Peterson et al. 1977). Higher temperatures cause a simultaneous increase in the rate of oxygen demand and a decrease in the solubility in water of atmospheric oxygen. Decreased temperatures generally cause slower growth and development, delaying the time to hatching (Koski 1975).

Changes in the intragravel water temperature and dissolved oxygen content in three Oregon coastal streams were monitored by Ringler and Hall (1975). They noted significant, although non-lethal, changes in these parameters resulting from reduced forest cover over the streams and increased deposition of fine sediment in the gravel. Urbanization may be expected to affect intragravel water quality in a similar fashion.

Considerable attention has been focused recently on declines in pH in natural waters resulting from an increase in acid-forming ions from anthropogenic sources. The effects of acidification have caused the disappearance of fish populations in numerous North American and European lakes (Beamish 1976, Almer et al. 1974). Several studies have documented lethal pH levels for salmonid embryos (Carrick 1979, Daye 1980, Trojnar 1977). Low pH apparently inhibits the hatching enzyme, chorionase, which may result in mortality or delayed hatching of salmonid alevins (Peterson et al. 1980).

Ammonia is present in natural waters in either the ionized (NH_4^+-N) or un-ionized (NH_3-N) form. The concentration of un-ionized ammonia, which is the form most toxic to fish, is dependent not only on total ammonia concentration, but also on pH, temperature, and ionic strength of the medium. NH_3-N levels increase with pH and temperature and decrease with an increase in ionic strength (Hillaby and Randal 1979). The specific biochemical mechanism of ammonia toxicity is unknown, but Rice and Stokes (1974) and Rankin (1979) have determined short and long-term median tolerance limits (TLm) for salmonid eggs and alevins. TLm estimates range from 0.059 mg $NH_3\text{-N}\cdot l^{-1}$ over 24 hours to 0.11 mg $NH_3\text{-N}\cdot l^{-1}$ for the entire period from fertilization to hatching (62 days).

As an end product of amino acid metabolism, ammonia is excreted by fish in its toxic form. Incubating salmonid eggs rely on the flow of surrounding water to dilute and transport ammonia away from the egg surface. In addition to the factors mentioned above, therefore, the concentration of un-ionized ammonia in intragravel water is influenced by the level of metabolic activity and by the flow rate of water through the gravel bed.

5.0 DESCRIPTION OF STUDY AREAS

5.1 Location and Characteristics of Study Areas

The two streams selected for study are Kelsey and Bear Creeks, located within King County, Washington, approximately 10 and 40 kilometers, respectively, west of Seattle. The region is a lowland area formed by glaciation, and the soils are a heterogeneous mixture of glacial and alluvial deposits (Banos and Bohrer 1972).

The Kelsey Creek watershed consists of 4 major subdrainages, with a total low flow surface area of 34,833 m^2 (Table 2). The main stem of Kelsey Creek arises from Larsen Lake and flows approximately 12 kilometers through Bellevue before discharging into Lake Washington (Fig. 6). The stream is small (1-5 m in width) and may be divided into three distinct reaches on the basis of gradient and general habitat characteristics. The upper reach lies between the outlet at Larsen Lake (Fig. 6) and the culvert at 148th Street and has a low gradient. Approximately 400 meters of the stream flows under the parking lot of a shopping center located at 148th and N.E. 8th Streets (Fig. 6). The remainder of the stream in this section is bordered by brush and hardwood vegetation, although the construction of a business park just north of the shopping center has recently commenced. Upstream movement of adult salmon and trout is denied by a chain-link fence across the stream at the 148th Street culvert. Salmonid habitat in this reach of stream is marginal.

The major portion of the stream used for spawning and rearing by salmonids extends from 149th Street to the lower end of Kelsey Creek Community Park, a distance of about 5 kilometers. Nine of the ten study areas sampled for fish

Table 2. Low flow surface area of Kelsey Creek and tributary stream areas accessible to anadromous fish.

Stream	Boundaries	Surface area (m^2)	Source
Kelsey Creek	Mouth to 148th N.E.	20,559	Richey (1982); Comis (1971)
Richards Creek	Mouth to Forks	3,837	Comis (1971)
West Tributary	Mouth to ∿ Bel-Red Road	5,556	Comis (1971)
Valley Creek	Mouth to ∿ N.E. 40th	4,881	Comis (1971)
	Total	34,833	

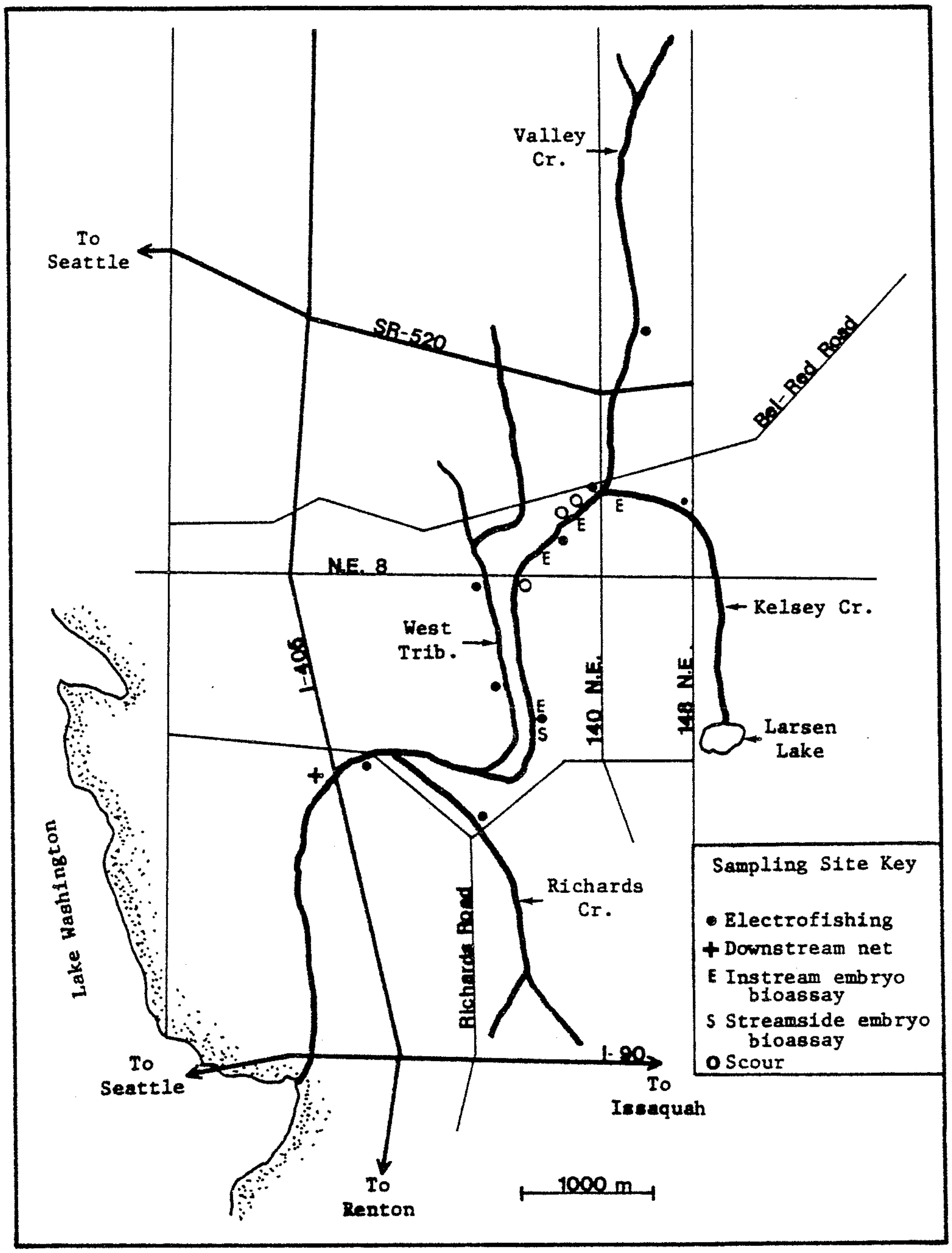

Figure 6. Kelsey Creek watershed and sampling sites.

were located in this segment of the stream. The gradient is moderate and although the stream has been altered considerably in the past, most development in this area has stabilized. Building and highway construction remains active along North Valley Creek, which joins Kelsey Creek at the intersection of 140th Street and Bel-Red Road.

The downstream reach of the urban stream flows through a low gradient slough which has a bed of mud and aquatic vegetation. Although it is unsuitable for salmonid spawning, there is some evidence that the lower section of the stream provides rearing habitat for juvenile salmonids. Consequently, one study area was established near the I-405 highway culvert in the same locality as the USGS gaging station (Figure 6).

At one time Kelsey Creek was a major producer of coho salmon (Oncorhynchus kisutch), and also supported significant numbers of cutthroat trout (Salmo clarki) and kokanee salmon (Oncorhynchus nerka). In a 1956 survey of Lake Washington streams, Ajwani (1956) indicated that the fisheries resources of Kelsey Creek were already in jeopardy from encroaching development within the drainage. More recently, Griggs (1972) obtained population estimates of resident cutthroat and coho juvenile in Kelsey Creek which suggested that the former species was the more abundant of the two. Kokanee populations have recently declined throughout the Lake Washington drainage in response to the successful introduction of sockeye salmon in major tributaries in the area. Kelsey Creek was not used by spawning sockeye salmon during this study. The City of Bellevue, the Municipality of Metropolitan Seattle, and the Washington State Department of Fisheries are vigorously pursuing a variety of stream rehabilitation programs, including the enhancement of salmon stocks through seeding and egg incubation projects, toxicant inventory, habitat restoration, public education, and surface water management plans.

The historical record of development of Bellevue typifies that of many other cities in the Pacific Northwest. Initial settlement of Bellevue occurred in 1883 when Isaac Bechtel settled on the shores of what is now Meydenbauer Bay. Hampered by its relative inaccessibility, the greater Bellevue area grew slowly, reaching a population size of only 400 by 1900 (Bellevue Planning Dept. 1977). Land at this time was used primarily for logging, with farming becoming increasingly important in the early 1900's. Population density continued to be low until the late 1940's (Fig. 7). A map of land use for the Bellevue area in 1936 (Fig. 8) shows almost the entire Kelsey Creek drainage basin to be undeveloped. Growth was stimulated in the late 1940's by several factors, the most important of which was the construction of the Lake Washington Floating Bridge. Following the removal of bridge tolls in 1949, drastic changes in land use began to occur, marked particularly by the appearance of low-density residential housing (Fig. 9).

From 1950 to 1970, the population of the greater Bellevue area increased by nearly 600 percent. Land use records for the Kelsey Creek watershed in 1959 reveal that residential housing occupied a substantial portion of land by that time, and a limited number of industrial plants had been constructed as well (Fig. 10).

Growth during the 1970's slowed, reflecting the depressed economy and the approach of saturation land development. The population in the City of Bellevue was estimated as 67,000 people in 1976 (Bellevue Planning Dept. 1977). Land use in the Kelsey Creek watershed is currently 54 percent single and

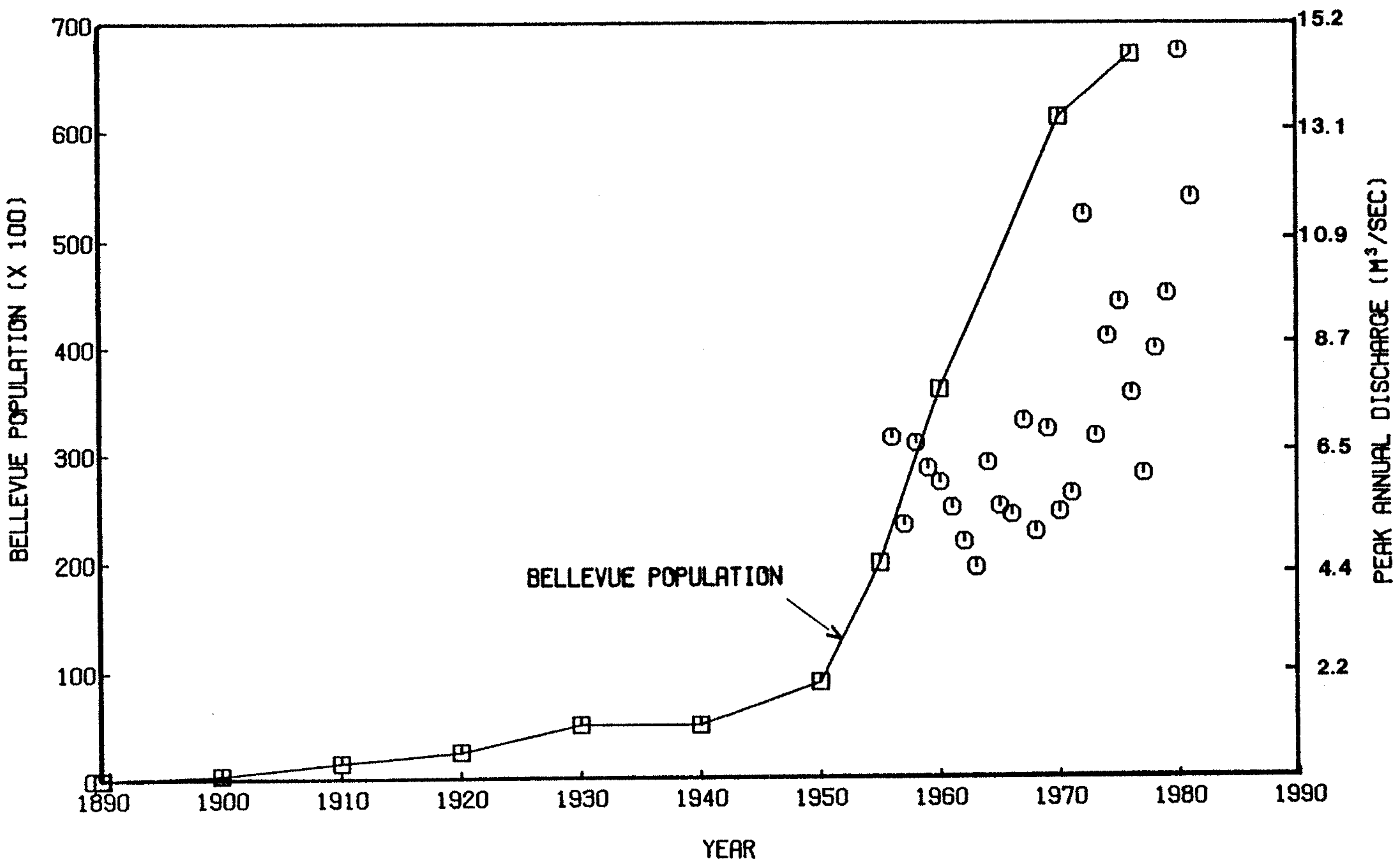

Figure 7. Population of Bellevue and peak annual discharge in Kelsey Creek (Ⓘ). Data from U.S.G.S. and Bellevue Planning Dept. 1977.

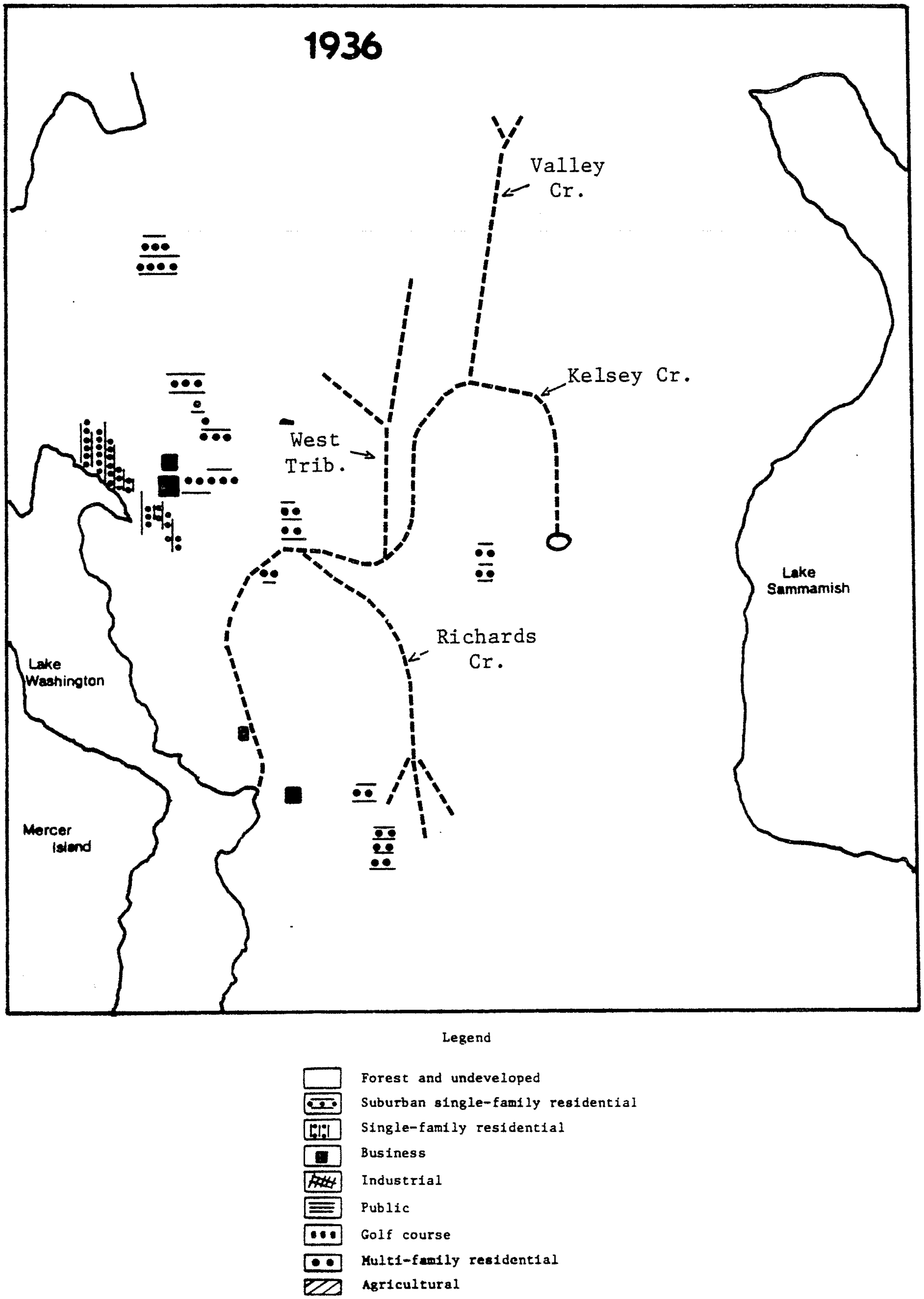

Figure 8. Land use in the Bellevue area, 1936 (redrawn from Walker 1961).

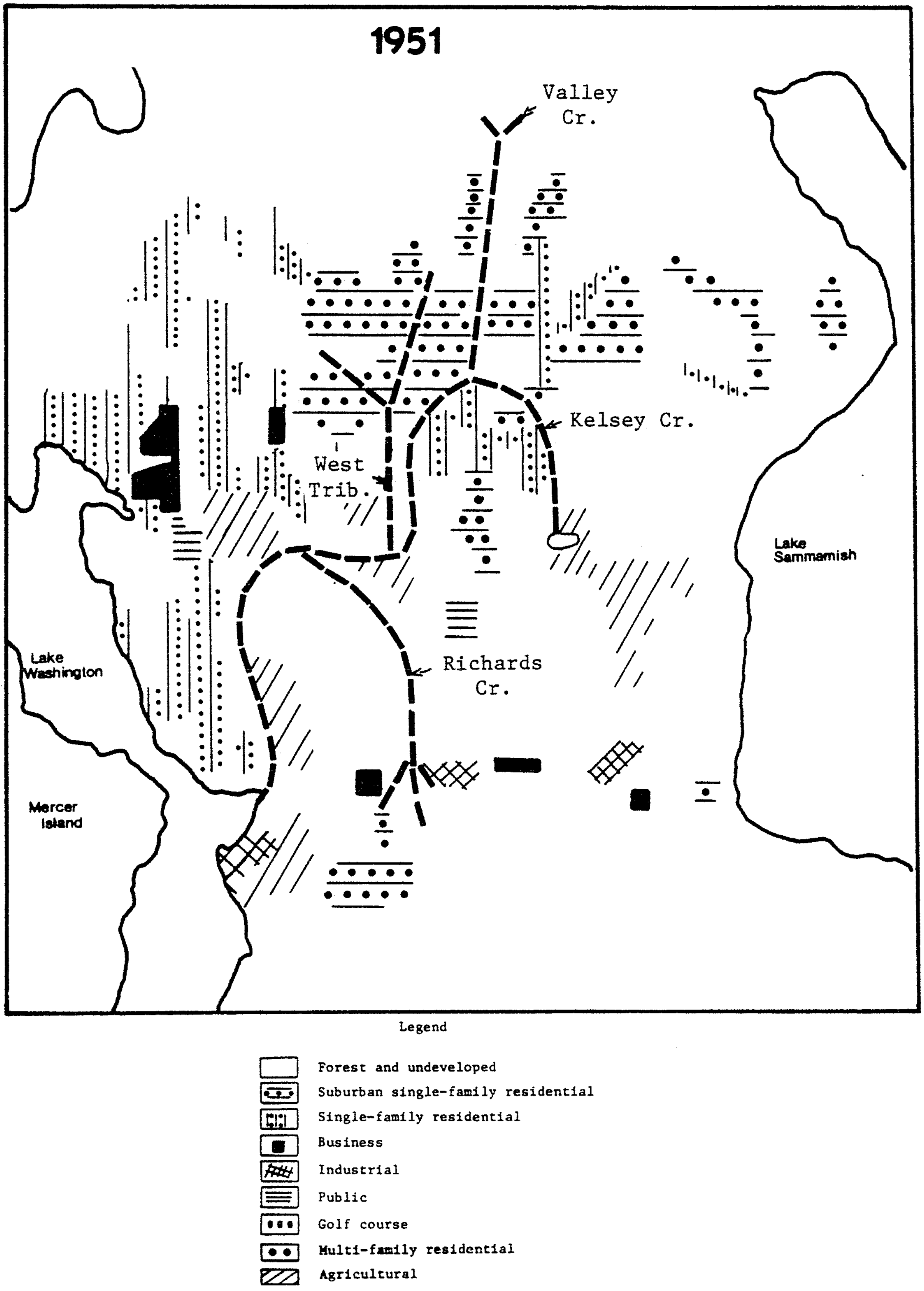

Figure 9. Land use in the Bellevue area, 1951 (redrawn from Walker 1961).

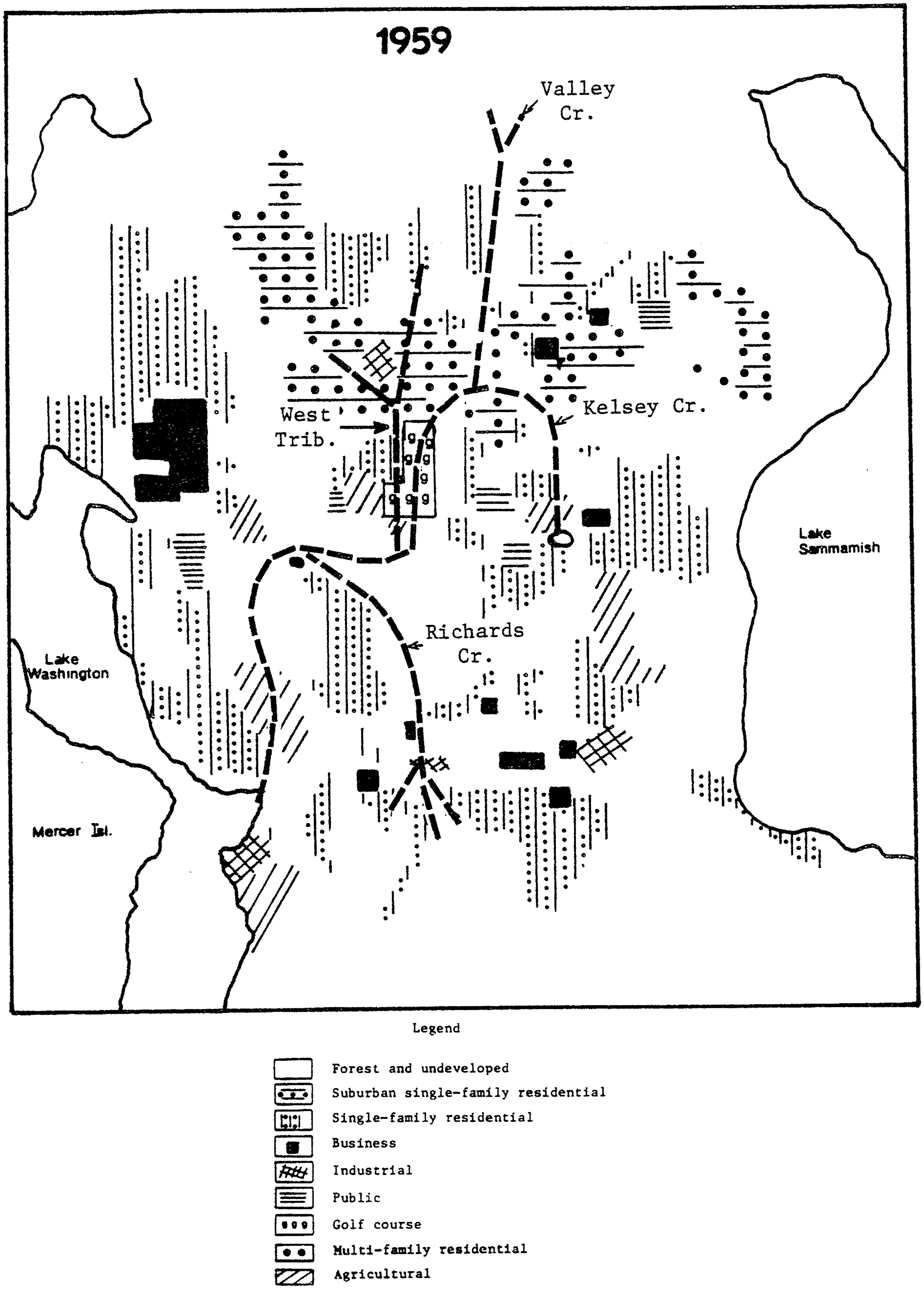

Figure 10. Land use in the Bellevue area, 1959 (redrawn from Walker 1961).

multifamily residential, 24 percent commercial and industrial, and 22 percent park and undeveloped (Perkins et al. 1980).

The hydrologic regime of Kelsey Creek has undergone a fundamental modification in recent years. Peak annual discharges have exhibited a general trend upward since 1970 (Fig. 11). This trend may be examined more precisely by plotting annual flood arrays from the periods 1956-1965 and 1971-1981. The curve generated by floods in the latter years (Fig. 11) lies distinctly above that for the period between 1965 and 1975. In fact, a flood which previously had a return period of 10 years may now be expected to occur at least every other year.

Although the trend of increasing peak annual discharges is consistent with observations from other urbanizing watersheds, it is not necessarily true that the alterations in land use which accompanied urbanization are the causative factor in this case. An alternative hypothesis is that the quantity of incoming precipitation has also increased during the last decade. If this explanation is correct, then annual flood arrays from other streams near Kelsey Creek should show a similar dichotomy between the test periods. Three streams (Big Soos Creek, Taylor Creek, and Newaukum) were chosen to test this alternative hypothesis based on the presence of an adequate water record, the absence of urban development within their watersheds, and their proximity to Kelsey Creek.

In contrast to Kelsey Creek, two of the three control watersheds (Taylor and Newaukum Creeks) displayed a reduction in the size of annual floods since 1971, and in the third (Big Soos Creek) no change was observed (Figs. 11 and 12). These conclusions were tested statistically with a Mann-Whitney U-test and were found to be valid in each case.

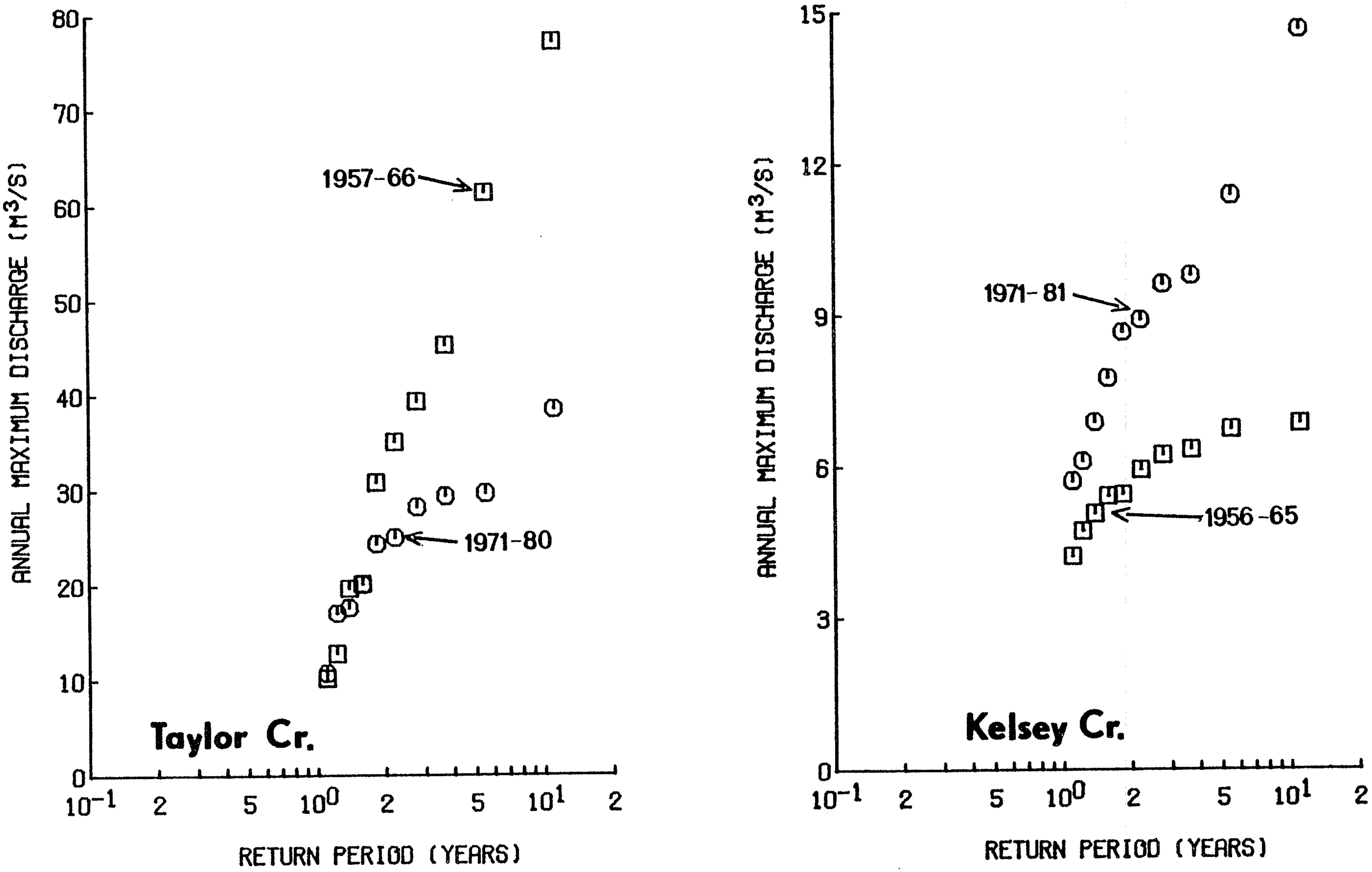

Figure 11. Annual flood return periods for Taylor Creek and Kelsey Creek (data source: U.S.G.S.)

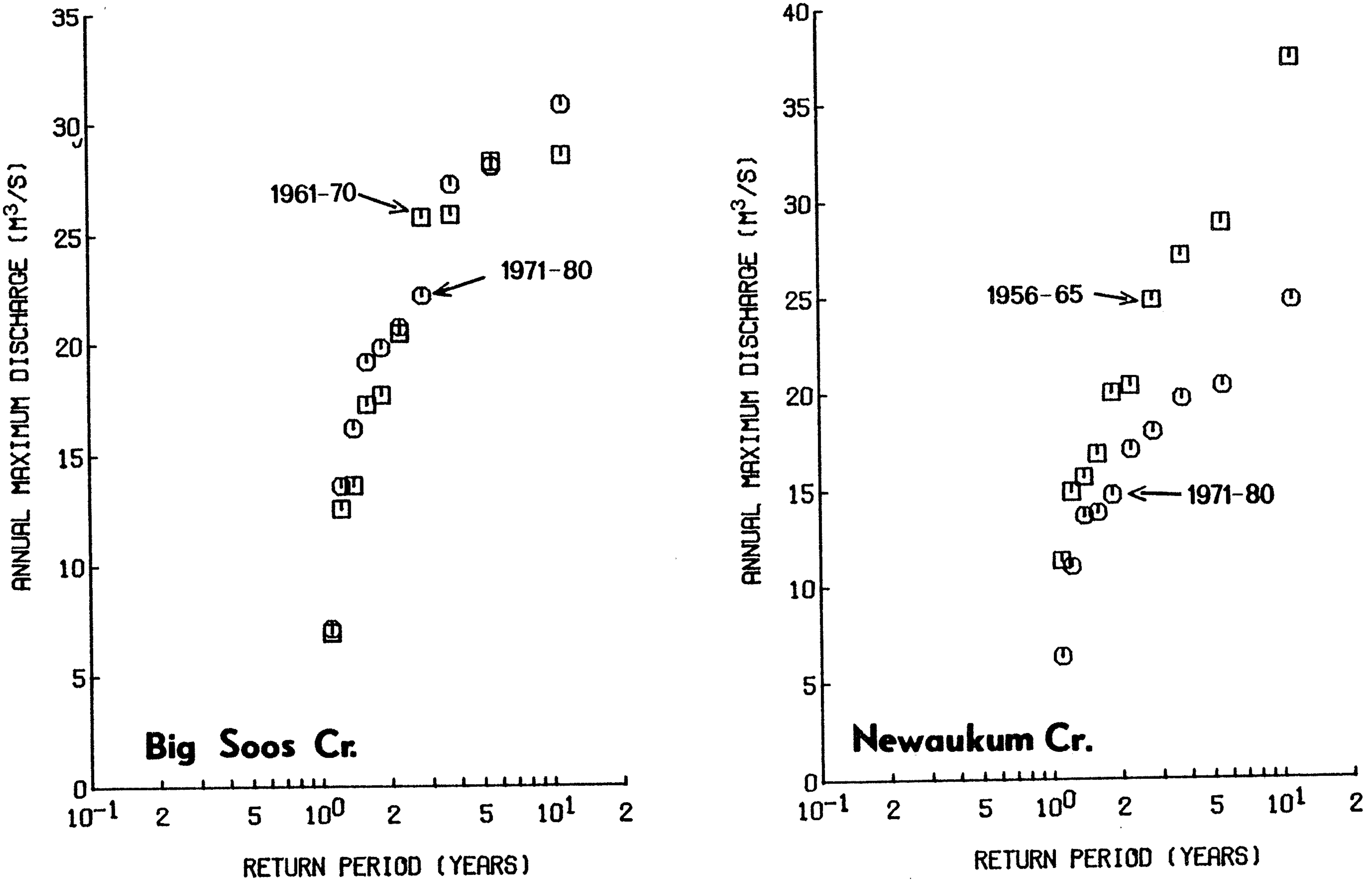

Figure 12. Annual flood return periods for Big Soos Creek and Newaukum Creek (Data source: U.S.G.S.)

The role of precipitation may be assessed more directly by establishing the relationship between it and stream discharge. National Weather Service records of rainfall at the Seattle-Tacoma Airport were collected and the amount of precipitation in the 15 hours preceding the peak discharge at Kelsey Creek determined. The peak discharges in "preurban" and "urban" years were then regressed upon the precipitation which caused them. The regression for "preurban" years was significant at an $\alpha \leq .025$ ($r^2 = .70$), and that for urban years at $\alpha \leq .01$ ($r^2 = .70$). The regression equations also differed significantly ($\alpha \leq .10$), (combined test for slope and elevation, Zar p. 320) with less precipitation required to produce a flood of a given magnitude during urban years (Fig. 13). It should also be noted that data from 1979 and 1980 were not included in this analysis. These points (Fig. 13) indicated that further alteration of the Kelsey Creek hydrologic regime may be occurring.

Owing to the extent of development within the Kelsey Creek watershed, upstream-downstream comparisons were not considered feasible as a means of evaluating the ecological effects of urban runoff. Alternatively, a stream in a nearby, comparatively undeveloped watershed was chosen to serve as a control stream (Perkins et al. 1978). Several physical and chemical characteristics of the two streams were assembled by Sloane-Richey et al. (1981) and are presented in Table 3.

Bear Creek discharges into the Sammamish River near the City of Redmond, Washington, and its characteristics are believed to be representative of those associated with streams draining undisturbed areas of the Pacific Northwest. Land use within the Bear Creek drainage, which is comparable in size (3600 ha) to Kelsey, is minimal; single-family residences comprise roughly 15 percent of the watershed, with the remaining 85 percent divided between

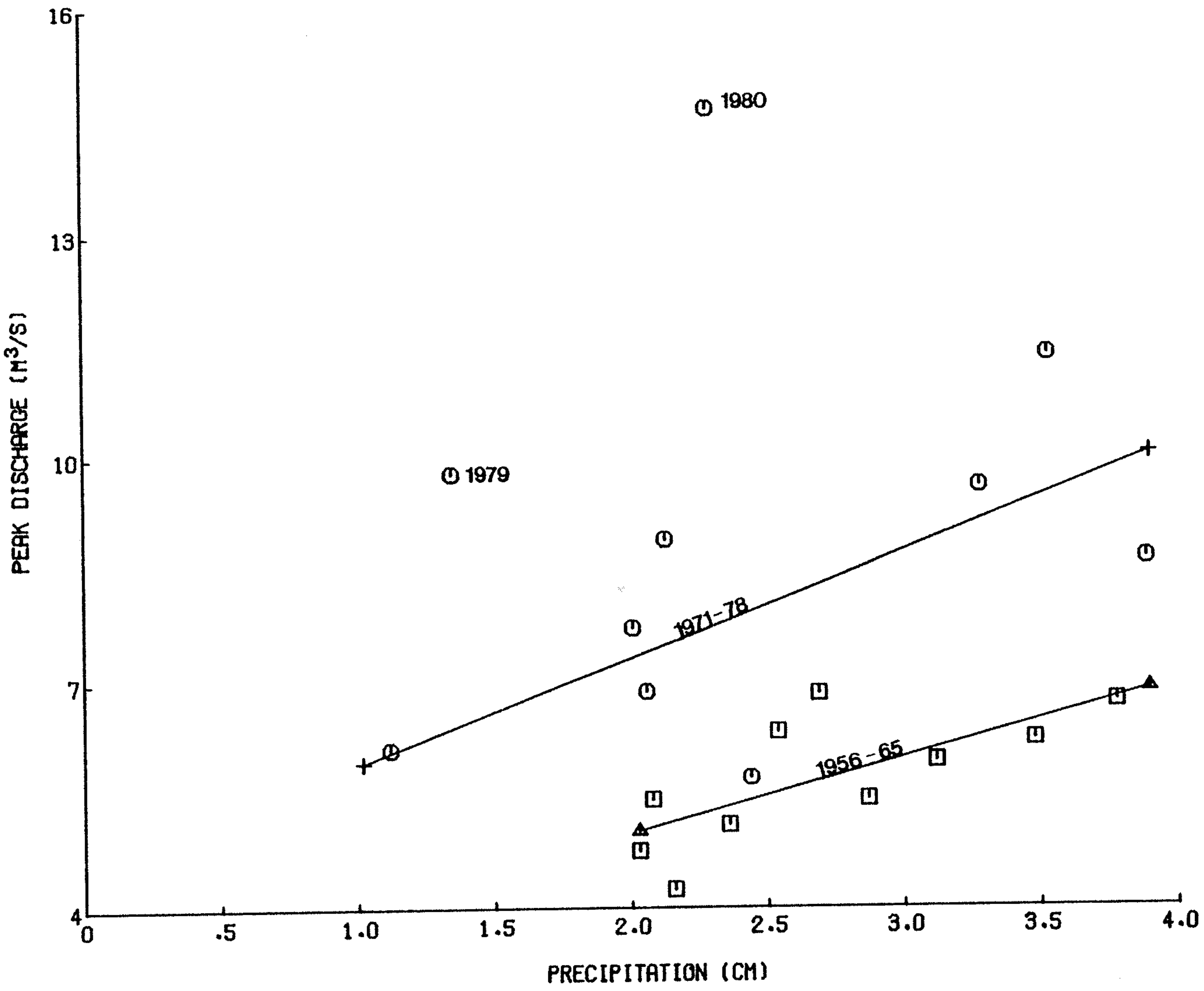

Figure 13. Relationship between the peak annual discharge in Kelsey Creek and the precipitation which preceded it for pre-urban (1956-1965), urban (1971-1978), and more recent (1979-1980) years.

Table 3. Physical characteristics of Kelsey and Bear Creeks (from Sloane-Richey et al. 1981).

	Kelsey Creek	Bear Creek
Drainage area (ha)	3109	3600
Maximum summer temperature (°C)	23.0	23.0
Minimum winter temperature (°C)	5.0	3.2
Median substrate size (mm)[1]	36.72 ± 6.84	27.5 ± 4.85
Maximum instantaneous discharge Dec. 1979 (m^3/s)[2]	8.68	6.31
Minimum instantaneous discharge July 1979 (m^3/s)[2]	0.20	0.13
Soluble reactive phosphorus (μg P/1)[1]	89 ± 5	25 ± 5
Nitrate + nitrite - nitrogen (μg N/1)[1]	722 ± 25	483 ± 104
Ammonium - nitrogen (μg N/1)[1]	±	±
Total suspended solids (mg/1)[1]	25 ± 4	8 ± 2
Total soluble lead (μg Pb/1)[1]	8 ± 1	4 ± 0

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[1] Data are presented as means for the sample period 1 standard error.

[2] Data are courtesy of the United States Geological Survey, Tacoma Office.

pasture and woodland (Sloane-Richey et al. 1981). Three study sites were selected in the upper reach of Bear Creek in areas of low gradient (Fig. 14).

Bear Creek has historically supported large populations of salmon and trout (Ajwani 1956). Fisheries records dating back to 1925 indicate that the stream was used heavily by kokanee as a nursery area. Coho (silver) salmon were introduced into Bear Creek in 1933 by the State Department of Fisheries and were abundant enough by 1956 to lead Ajwani (1956) to state: "For its size, this stream is probably the heaviest producer of silver salmon, and one of the largest producers of silver trout (kokanee) in the state." At present there are large runs of sockeye and coho salmon in the stream, a fair abundance of cutthroat trout, and very few steelhead and chinook salmon.

Water records for Bear Creek are not available for years prior to 1979. Data collected since that time indicate that the hydrologic regime of Bear Creek is representative of the streams draining rural areas of Western Washington (Perkins et al. 1980).

5.2 Previous Research in Study Areas

Information on the physical, chemical, and biological characteristics of the study streams is available from a variety of published and unpublished sources. Since Kelsey and Bear Creek are subdrainages of the Lake Washington watershed, much of the information is presented in conjunction with monitoring programs of regional feeder streams. At least four major studies, however, have been completed on Kelsey Creek, and a fifth is nearing completion. Bear Creek, by comparison, has received little attention with the exception of the present study.

Kelsey Creek was the subject of an ambitious interdisciplinary study

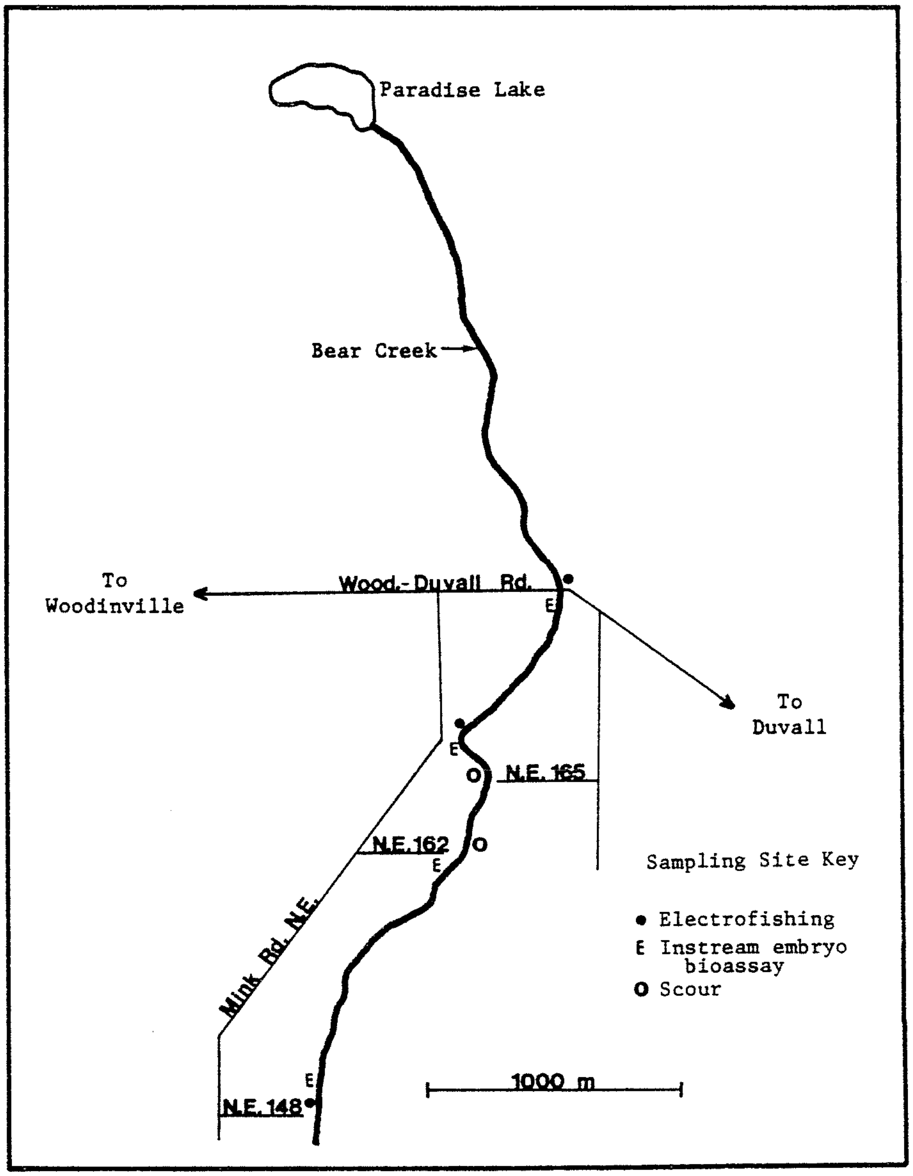


Figure 14. Bear Creek watershed and sampling sites.

conducted by Comis et al. (1971) which sought to identify and measure the environmental response of the stream to urban development within the watershed. The project was organized into four major working sections: physical, biological, water quality, and applied. The study concluded that the total volume of runoff, expressed as a monthly average, had increased only slightly over pre-urbanized years, but that flood peaks have increased two to three times over the same time period. The increase in the intensity of runoff had a measurable effect on the channel stability of Kelsey Creek. In regard to water quality parameters, it was determined that coliform bacteria, dissolved oxygen concentrations, and the biological oxygen demand in the stream had not yet reached deleterious levels.

Of particular interest to Comis et al. (1972) were the observations relating to the juvenile fish populations and their associated habitat in Kelsey Creek. Cutthroat trout were found to be the dominant salmonid species, accompanied by low numbers of coho salmon and an occasional chinook salmon. The condition factors of the sampled fish were comparatively high, implying that food availability was not a limiting factor. During the course of the study a newly installed culvert at the lower end of Kelsey Creek was determined to block upstream fish passage under certain flow conditions and, after notification of the WDF, the problem was rectified. A major factor thought to impair salmonid reproduction in the urban streams was siltation resulting from construction activities.

In contrast to the broad scope of the study by Comis et al. (1972), the River Basin Coordinating Committee (RIBCO), established in 1970 by METRO, selected Kelsey Creek as one of several "demonstration streams" for evaluation of specific hydrological problems associated with urban stream runoff. The result of this study was the recommendation of two alternative plans for drainage control in the watershed in order to ameliorate problems associated with further urban development. The first alternative called for the modification of drainage facilities, including riprapping the natural stream channels to pass peak flows predicted from future load use projections. The second option, judged the more practical by the committee, recommended the provision of in-channel detention reservoirs and on-site runoff control structures for peak flow attenuation.

Conley (1974) reviewed RIBCO's proposals and the study by Comis et al. in his analysis of the effects of urbanization and storm and surface water management practices on Kelsey Creek. He concluded that a more vigorous investigation of the ecology of the stream was unnecessary and that future efforts be directed toward administering legal controls necessary to prevent further damage to the natural drainage system.

As part of its Areawide Water Quality Study, METRO has monitored water quality parameters in King County streams over the past decade. Much of the data has been compiled and summarized in Environmental Management for the Metropolitan Area Studies, Part III — Water Quality (Stevens et al. 1974), and in Technical Appendix No. 5: A Profile of Water Quality in the Cedar-Green River Basins (1978). Problems observed in Kelsey Creek were high water temperature and fecal coliform counts, although the latter index varied considerably throughout the drainage system. Bear Creek was distinguished by high fecal coliform counts and high inorganic nitrogen and total phosphorus concentrations. Monitoring of water quality is being continued on both streams and periodic updates are published by METRO as

Implementation Progress Reports.

METRO also funded a baseline study of the benthic macroinvertebrate communities in King County streams, the results of which are available in Technical Report No. 12. The authors used four index parameters (mean total number, mean total volume, number of taxonomic groups, and the percentage composition of Ephemeroptera, Plecoptera, and Trichoptera insect taxa) to assign each stream to a "poor," "fair," and "good" quality category. Kelsey Creek was judged to be poor in quality and ranked lower in all index parameters than Bear Creek, which was considered to be of fair quality.

The Resource Planning section of the King County Planning Division analyzed data from METRO, USGS, and the Departments of Fisheries and Game sources in an attempt to establish a cause-effect relationship between urban development and stream degradation for 15 King County streams (King County Planning Division 1980). Variables describing the percent of drainage basin covered with impervious surface, peak flows, water quality, aquatic insects, and salmonid escapement were used to rank the streams. Bear Creek ranked 3 in percent impervious surface, 1 in peak flow, 5 in water quality, 6 in aquatic insects, and 2 in salmon escapement. Kelsey Creek ranked 12, 8, 15, 15, and 8 in the same categories, respectively, indicating less desirable conditions.

The ecological impacts of runoff, including the fisheries-related aspects which are detailed in this report, consisted of research into the physical, chemical, and biological characteristics of Kelsey and Bear Creeks as each was affected by urbanization. A considerable amount of information pertaining to the water quality, periphyton, and macroinvertebrates of Kelsey and Bear Creeks has not been referred to in the text and is available from Dr. Michael A. Perkins of the Department of Civil Engineering of the University of Washington.

Results of the investigation of water quality aspects of the study streams indicate 1) increased nutrient concentrations in Kelsey Creek relative to Bear Creek, 2) the suspended load of particulates typically runs one to two times greater in Kelsey than in Bear Creek, 3) inorganic silt comprises the dominant fraction of the suspended particulate load in Kelsey Creek, and 4) concentrations of water-borne toxicants in the study streams appear to be negligible.

In regard to the periphyton communities of Kelsey and Bear Creeks, it was determined that 1) the mean standing crop of particulate organic matter was similar in the two streams, but that 2) there is a significant difference in its quality as food for higher trophic levels and the timing of its availability. Sloane-Richey et al. (1981) demonstrated that organic accumulations in Kelsey Creek are comprised predominantly of periphyton during much of the year whereas allocthonous detrital material dominated in Bear Creek. The lower food quality of the particulate organic matter found in Kelsey Creek resulted from increased siltation, rapid downstream transport of detritus, and a lack of riparian vegetation.

Aquatic macroinvertebrates which depend upon grazing, particle feeding, and filtering, or whose life cycles are dependent upon autumnal inputs of allocthonous material are comparatively rare in Kelsey Creek. As might be expected, the urban stream is dominated by invertebrates such as amphipods and oligochaetes which appear capable of tolerating rapid fluctuations in food availability and flow regimes, decreased food quality, and the loss of interstitial habitat due to siltation. Pedersen (1981) showed that the functional diversity of macroinvertebrates in Kelsey Creek was very low relative to Bear Creek; the latter stream supports a benthic community represented by a wider array of functional groups from the Ephemeroptera, Plecoptera, and Trichoptera orders of aquatic insects.

The City of Bellevue, METRO, and the USGS are currently working together under a cooperative agreement to characterize the water quality of urban runoff in the Kelsey Creek watershed. Most of the research has focused on identifying sources of urban runoff and monitoring their concentrations in the runoff from separate residential catchments in Bellevue. METRO has also collected samples from surface and interstitial water in Kelsey Creek as part of their Toxicant Inventory Analysis. The samples are being analyzed for a total of 129 EPA-designated "priority" pollutants. Progress during the first year of the various projects is summarized in Bellevue Urban Runoff Project and Pitt et al. (1981).

6.0 MATERIALS AND METHODS

6.1 Gravel Sampling and Analysis, 1979-80

Several problems relating to substrate quality were investigated during the initial year of the study. The first concerned the temporal trends in streambed composition occasioned by seasonal changes in runoff. It was hypothesized that streambed features vary over time in response to discharge patterns, particularly during the winter high-flow periods. The redd construction activities of spawning fish were also thought to have some effect on bed composition. A second problem dealt with the spatial variability in the substrate which may occur along the length of the stream. It seemed likely that the bed characteristics in the upper reaches of the channel would differ from those in downstream sections. The third problem addressed differences in bed composition in streams draining urban and rural watersheds to determine the impact of land development on the quality of spawning habitat. Two gravel composition parameters were evaluated, the percentage of fines less than 0.841 mm and the mean geometric diameter, in relation to intragravel oxygen levels. A final topic of interest was the effect of hydrologic and hydraulic variables on streambed composition.

Substrate samples were collected from Kelsey and Bear Creeks at the field stations identified in Figures 6 and 14, respectively. Study site locations and designations were as follows:

KC1 — Kelsey Creek, extending downstream from approximately 50 m below the 148th N.E. bridge;

KC2 — Kelsey Creek, extending downstream from approximately 20 m below the 140th N.E. bridge;

KC3 — Kelsey Creek, extending upstream from approximately 30 m above the 134th N.E. bridge;

KC4 — Kelsey Creek, within the Kelsey Creek Park;

KC5 — Kelsey Creek, extending upstream from the railroad trestle near Interstate 5;

BC1 — Bear Creek, extending upstream from approximately 10 m above the Woodinville-Duvall Road;

BC2 — Bear Creek, extending upstream from the foot of N.E. 167th Pl.;

BC3 — Bear Creek, extending upstream from the foot of N.E. 148th.

Gravel samples were taken from areas selected to represent the optimal spawning habitat in each of the stations established on Bear and Kelsey Creeks. All of the sampling areas on Bear Creek were subsequently utilized by spawning salmon and in Kelsey Creek three of the five gravel areas (KC2, KC3, and KC4) were used by spawning cutthroat trout. Spawning was not observed at stations KC1 and KC5.

Gravel samples were obtained on four dates separated by three-month intervals using a manually operated core sampler, commonly known as a McNeil sampler (McNeil and Ahnell 1964). Three replicate samples were removed from the center of the channel approximately 0.5 m apart. This method of sampling was justified since the bed material appeared more homogenous in the center of the channel and less variable along the length of the stream rather than in a lateral direction. Care was taken to avoid disturbing the stream prior to sampling. The circular sampler removes a core 15.24 cm in diameter and 20 cm deep. A plunger was used to capture suspended sediment in the tube after the coarse material had been removed by hand. All samples were collected by a single individual to avoid operator selectivity. Individual samples were placed in 5-gallon buckets lined with plastic bags and returned to the laboratory for analysis. The contents of each container were washed through a series of 10 Tyler screens which effectively separated the particle size groups. The screens had the following square mesh openings: 50.8, 26.9, 13.5, 6.73, 3.36, 1.68, 0.841, 0.420, 0.210, and 0.105 mm. Material passing through the smallest sieve was concentrated in a large funnel and allowed to settle for approximately one hour. The volume of solids collected in a graduated cylinder at the base of the funnel was measured to the nearest 10 ml. For the purposes of this study it was assumed that the fine-grained material collected in the cylinder averaged 0.063 mm in diameter, the size class known as the "wash load" of channel sediments (American Geophysical Union 1947). The volume of material remaining on each screen was measured to the nearest millimeter using a volumetric device developed by McNeil and Ahnell (1964).

The volumetric displacement method of determining bed composition has gained wide acceptance in studies concerned with the quality of salmonid spawning habitat (Lotspeich and Reid 1980). The basic assumption underlying the method is that the porosity and sphericity of the particle volume retained on each sieve are about the same for each particle size group (Young and Wiskus 1979). The measurement of size and gradation of substrate material by volume has been advocated as a simple, rapid, and accurate substitute for the gravimetric method.

The data collected by the volumetric method was corrected for the bias resulting from the increased water-holding capacity of finer sediments. Following the suggestion of Shirazi and Seim (1979), the dry contents of the 0.210 mm sieve were used to estimate the density of the substrate by dividing the dry weight of the sample in grams by the volume of water it displaced in cubic centimeters. The density was estimated for two samples from each stream on each sampling date. After averaging, these estimates enabled correction factors to be applied to volumetric data in order to derive dry weight estimates of the different particle size classes.

Following conversion to dry weights the data were analyzed by a least squares linear regression of the logarithmic transformation of size class diameter (i.e., mesh size) against the inverse probability transformation of a variable representing the percentage of the sample finer than the size class in question (Shirazi and Seim 1979). A high correlation coefficient was interpreted as a test for lognormality of size class distribution. If the distributions are found to be lognormal then the regression procedure reduces the variability inherent in using the untransformed data. It also facilitates an analysis of the entire textural composition of the gravel sample and enables calculation of appropriate statistical parameters. Several studies have used the lognormal distribution to characterize substrate materials (Vanoni et al. 1961, Renard 1974, Platts et al. 1979, Shirazi and Seim 1979).

The computer program SEDIMENT (FRG-367), written by Gales and Swanson (1980), was used to perform the operations described above. The program generated a table for each set of replicates containing the following statistics: 1) the slope, intercept, and correlation coefficient of the regression line; 2) a series of particle diameters in millimeters corresponding to the 5th (d_5), 16th (d_{16}), 50th (d_{50}), 84th (d_{84}), and 95th (d_{95}) percentiles, calculated from the inverse of the regression line; 3) an estimate of the geometric variance, σ_g, which corresponds to the ratio of d_{84}/d_{16}; and 4) the percentage of fine sediment less than 0.841 mm in diameter. The program optionally plots a a scattergram of the values for each regression line, and a table of "percent finer" values which represent the percentage of the total adjusted volume collected by the sieves in a given replicate which trap finer (smaller) sediments. The algorithms used in SEDIMENT are based on methods described by Shirazi and Seim (1979).

When the particle distribution in a gravel sample is lognormal, the geometric mean of the distribution, d_g, is equal to the median particle size, d_{50} (Platts et al. 1979). The percentage of fines less than 0.841 mm, d_g and σ_g are convenient statistical measures which relate to the porosity and permeability of the gravel. A direct relationship between the survival of salmonid embryos and the permeability of the spawning bed has been demonstrated (Wickett 1958, Cooper 1965). Thus, these parameters are potentially adequate measures of substrate quality as it affects embryo survival.

The usefulness of being able to predict percent fines knowing d_g, and vice versa, is an issue raised by Platts et al. (1979). A regression of the logarithm of percent fines values against the logarithm of d_g values was carried out as a means of evaluating the strength of association between two variables.

6.1.1 Profile Analysis

The gravel composition data collected on four dates from the study streams may be represented graphically as a plot of the means and standard deviations of the data. Plotting the mean percent fines or d_g values for Kelsey and Bear Creek results in a set of profiles which may be interpreted as bed composition responses to environmental changes over time. A preliminary inspection of the data suggested that a test be made for differences in the levels and in the shapes of the stream profiles. It was hoped that this would enable some inferences concerning the temporal variability in bed composition and possible differences between the two streams.

Profile analysis is a multivariate type problem; in this study multiple measurements were obtained for gravel samples taken from two streams. The measurements, in the form of percent fines or d_g values, can be assumed to be observations on continuous random variables with some multivariate distribution with an arbitrary variance-covariance matrix. There are three questions which were asked concerning the stream profiles:

1. Do the profiles have the same shape, in the sense that the line segments of the adjacent streams are parallel? This is equivalent to the hypothesis of no stream by date interaction and is made clear by reference to the respective mean vectors of the variable percent fines (or d_g), represented as X. For the two streams, the mean vectors will be

$$\bar{X}_K = [\bar{X}_{K1}, \bar{X}_{K2}, \bar{X}_{K3}, \bar{X}_{K4}] \quad \text{and} \quad \bar{X}_B = [\bar{X}_{B1}, \bar{X}_{B2}, \bar{X}_{B3}, \bar{X}_{B4}]$$

If the corresponding differences between the stream means for each date are denoted by

$$\bar{X}_{K1} - \bar{X}_{B1} = d_1, \ldots, \bar{X}_{K4} - \bar{X}_{B4} = d_4,$$

then if $d_1 = d_2 = d_3 = d_4$, the profiles must be parallel. The null hypothesis of parallelism can be written in matrix form as

$$H_0: C\mu_K = C\mu_B$$

where C is the 3 x 4 transformation matrix. The segments are estimated for Kelsey Creek by

$$\bar{X}_K C' = [\bar{X}_{K1} - \bar{X}_{K2}, \bar{X}_{K2} - \bar{X}_{K3}, \bar{X}_{K3} - \bar{X}_{K4}]$$

and their sample covariance matrix is CSC'/N_K. S is the within-groups covariance matrix, and N_K is the number of sample means for Kelsey Creek. The estimates for the Bear Creek profile segments are similarly derived. The number of sample means for Kelsey and Bear Creek were 12 and 7, respectively, which are the number of samples obtained on June 26, 1979. Matrix computations require a uniform number of observations across dates; thus, for three subsequent dates, samples were randomly deleted from the stations which had missing values in June. The means and standard deviations of the original data and the data used in profile analysis will be presented for comparative purposes. The general position and shape of the profiles remained unchanged after the samples were deleted.

The two-sample T^2 statistic is used for testing the parallelism hypothesis and is computed from the differences of the four successive responses:

$$T^2 = \frac{N_K N_B}{N_K + N_B} (\bar{X}_K - \bar{X}_B)'C'(CSC')^{-1} C(\bar{X}_K - \bar{X}_B)$$

and the F statistic is

$$F = \frac{N_K + N_B - 4}{(3)(N_K + N_B - 2)} T^2$$

The hypothesis that the slopes of the profile segments are the same for both streams was accepted if the critical F value with 3 and 16 degrees of freedom exceeded the observed F statistic at the $\alpha = 0.10$ significance level.

2. The second question concerning profiles asks if the streams are on the

same level, i.e., do the separate profiles represent different populations or do they arise from populations having the same means on any given date? This refers to the hypothesis of equal stream effects and requires that no stream by date interaction be present. In other words, the test for parallelism must be accepted prior to testing for equal levels.

The null hypothesis for question 2 may be written as

$$H_0: \; j'\mu_K = j'\mu_B$$

where $j = [1,1,1,1]$ is a four-component vector with unity in each position. The two-sample t-statistic is computed as

$$t = \frac{j'(\overline{X}_K - \overline{X}_B)}{\sqrt{j'Sj(1/12 + 1/7)}}$$

If $|t|$ was less than the tabulated t with 17 degrees of freedom ($\alpha = 0.10$), then the hypothesis was accepted. It was assumed that two-sided t-rules applied.

3. The final question which may be asked is whether the response means of the two streams differ from date to date. If this is the case, then the profiles would be "flat," that is, they would be straight lines with a slope of zero. Again, the assumption of parallelism must be tenable. The hypothesis is written in matrix form as

$$H_0: \; C(\mu_K + \mu_B) = 0.$$

A grand mean vector,

$$\overline{X} = (12/19)\overline{X}_K + (7/19)\overline{X}_B ,$$

is used to compute the single-sample T^2 statistic

$$T^2 = (N_K + N_B)\overline{X}'C'(CSC')^{-1} C\overline{X}$$

If the profile slopes are equal to zero, then

$$F = \frac{17}{(17)(3)} T^2$$

has the F distribution with 3 and 17 degrees of freedom and we accept at $\alpha = 0.10$ the null hypothesis when the observed F is less than the critical value F.

Morrison (1976) and Winter (1971) have discussed the tests used in profile analysis and several applications to research in the social sciences. A general treatment of profile analysis as an additive analysis of the variance model is given by Greenhouse and Geisser (1959).

6.1.2 Analysis of Variance

The percent fines and d_g data can be grouped according to two factors potentially representing separate sources of variation: different sampling locations and different dates on which the samples were taken. A two-way analysis of variance (ANOVA), in which the factors were assumed to be fixed, was performed to test whether a significant portion of the variation in the data could be accounted for by at least one of the factors. If significant effects exist for both factors then we may infer that at least one of the sampling sites and at least one of the dates is different from the other locations and dates.

A common and often vexing problem in two-way ANOVA can be the presence of interaction, which may be considered a dependence of the effect of one factor on the level of another factor. An important assumption of this procedure is that if a given sampling site and date each contribute to a change in the bed composition, these two contributions add their effects without influencing each other. Interaction between factors is almost always present; however, it is not necessarily significant. When it is significant, it indicates that the effects of sampling site and date are not simply additive but that the effect due to the joint influence is either greater or less than the effects of each factor considered separately.

Regardless of the outcome of the preliminary two-way ANOVA, it was of interest to determine what specific differences exist among the group means. For example, if the four means calculated for the sampling dates are being compared and the two-way ANOVA reveals that a significant effect is present (i.e., H_0: $\mu_1 = \mu_2 = \mu_2 = \mu_4$ is rejected), we wanted to determine which subgroups of means was different. This was accomplished by considering more specific hypotheses such as H_0: $\mu_1 = \mu_2$, H_0: $\mu_2 = \mu_3$, H_0: $\mu_3 = \mu_4$, or even H_0: $\mu_1 + \mu_2/2 = \mu_3 + \mu_4/2$. Obviously, a large number of comparisons are possible.

The statistical procedure of testing for differences among means is the one-way ANOVA, in which only one independent variable is involved. Where only two samples are compared, the t-distribution is traditionally used to test significant differences between means. However, the analysis of variance is better suited for our purposes since it is a more general test and is mathematically equivalent to corresponding t-tests. _A priori_ tests were used for comparison among means in the comparatively few situations where differences were expected before data collection and analysis. For example, it was hypothesized that bed composition would differ during winter and summer in both streams. Another example is a test for differences between upper- and lowermost sampling sites on the same stream. Comparisons which suggested themselves after the data were collected were made using a posteriori tests. These comparisons are valid only if the preliminary overall ANOVA is found to be significant (Sokal and Rohlf 1969). Scheffe's test was used for a posteriori comparisons because it uses a single-range value for examining all linear combinations of group means. It is exact even for unequal group sizes, and is stricter than most other a posteriori tests (Kim and Kohout 1975). The SPSS subprogram ONEWAY was used to facilitate data analysis.

6.1.3 Multiple Regression

The objective of this part of the study was to relate site-specific fluvial characteristics in both streams to the gravel composition parameters, percent fines and mean geometric diameter. The replicate observations recorded for a given station and date were combined and averaged to reduce the variation in these parameters caused by longitudinal variability within the sampling area. Eight independent variables (Table 4) believed to influence the percent fines and mean geometric diameter of gravel samples were recorded for each composite sample. A multiple linear regression procedure was applied to analyze the data set.

The analytical framework of multiple regression techniques requires explanation for inference-making purposes if appropriate references are to be understood. In the present analysis, the channel characteristics (including bed composition) are considered to be random variables with an assumed multivariate distribution. This implies that a given dependent variable x

Table 4. Regression variables for Kelsey and Bear Creeks. See text for further details.

			Description
			Dependent
1.	FINES	:	Percent of sample < 0.841 mm in diameter
2.	D_9	:	Mean geometric diameter (mm)
			Independent
3.	Q	:	Mean discharge during preceding three months (cfs)
4.	MAXQ	:	Maximum discharge during preceding three months (cfs)
5.	MINQ	:	Minimum discharge during preceding three months (cfs)
6.	INSTQ	:	Instantaneous discharge (cfs)
7.	PRECIP	:	Number of rain-free days prior to sampling (days)
8.	GRAD	:	Hydraulic gradient(%)
9	DIST	:	Distance of sampling station from lake outlet (mi)
10.	SINU	:	Sinuosity of station at which samples were taken

$$F\ (y_1x_1,\ x_2 \ldots x_k) \quad = \quad \text{Prob}\ (Y \leq y,\ X \leq x_i,\ X_2 \leq x_2,\ \ldots\ X_k \leq x_k) \qquad (1)$$

can be accurately determined if all of the factors affecting it are known. It is unlikely that this condition is met and for our purposes a more useful distribution is the conditional distribution

$$G\ (y/x_1,\ x_2,\ \ldots,\ x_k) = \text{Prob}\ (Y \leq y \mid X_1 = x_1,\ X_2 = x_2,\ X_k = x_k)$$

where channel characteristics are identified by the sample variables x_1, x_2, ..., x_m, and either percent fines or d_g are given by y, then the conditional distribution allows the determination of the probabilities of variable levels of Y. Therefore, the problem of characterizing the relationship between channel characteristics and bed composition becomes one of determining the properties of parameters of the conditional distribution.

If the individual y observations are normally distributed and are statistically independent of one another, the mean value of Y for each specific combination of X_1, X_2, ..., X_m is given by

$$\mu_{Y/X_1,\ X_2,\ \ldots,\ X_k} = B_0 + B_1X_1 + B_2X_2 + \ldots + B_kX_k$$

$$y \quad = \quad B_0 + B_1X_1 + B_2X_2 + \ldots + B_kX_k + E$$

where E is the error parameter reflecting the difference between the observed mean Y and the true mean $\mu x/x_1,\ x_2,\ \ldots,\ x_k$. The variance of Y equals σ^2 and is the same for any fixed combination of X_1, X_2, ..., X_k.

The least squares estimate of the parameters μ and σ^2, of the conditional distribution is made on the basis of independent samples of channel characteristics. The best-fitting model chosen is that model which minimizes the sum of squares of the distances between observed and predicted responses. The fitted regression model is denoted by

$$y = b_0 + b_1x_1 + b_2x_2 + \dots + b_kx_k$$

in which x_1, x_2, ..., x_k are now the sample values for variables x_1, x_2, ... x_k. Similarly, the coefficients b_0, b_1, ..., b_k are estimates of B_0, B_1, ..., B_k. The mean square residual term provides an estimate of σ^2 under the assumed model.

One operational approach to the selection of an appropriate model is the use of a stepwise regression procedure which Draper and Smith (1966) believe to be the best of the variable selection procedures. This approach enters into regression. The X variables most highly correlated with the dependent variable, one at a time. At each step a variable is added if it results in a significant improvement in the explained variation of Y. A partial F test is performed for each variable already in the model, treating it as though it were the most recent variable entered, and the variable with the smallest non-significant partial F statistic, if present at all, is removed. The removal of a variable occurs if the entered variable is correlated with other variables already in the equation. A new regression equation is determined at the end of each step and the whole process continues until no more variables can be added or deleted. The final result should be an equation with a minimal number of uncorrelated channel characteristics as independent variables.

Kleinbaum and Kupper (1978) provide a detailed discussion of multiple regression techniques and appropriate significance tests. The general problem of selecting a suitable regression equation has been described by Draper and Smith (1966). The SPSS subprogram REGRESSION (Cohen et al. 1979) performed the necessary data analysis.

With the exception of PRECIP, the variables listed in Table 4 are specific for each sampling site. Discharge was measured at 12 to 15 transects at each station during summer low-flow periods. The discharge observed at a station upstream from the USGS gage was usually a fraction of the discharge recorded on the same date at the gaging station. Therefore, the average discharge measured in the field for each station was divided by the USGS geographical discharge of the same date to provide a correction factor. Average, minimum, and maximum discharges were determined from the USGS data for a one-month period preceding the sampling date, and then multiplied by the correction factor to determine site-specific values. The instantaneous discharges for the sampling dates were obtained in a similar manner.

PRECIP is a variable representing the number of rain-free days prior to sampling. Precipitation data were obtained from records provided by the Lake Union weather station.

Sinuosity was defined as the ratio of actual stream distance to the total straight line stream distance. The measurements were taken from a map of each station drawn from surveying data. Actual stream distance was measured from the center of the first transect to the middle of the last. Total straight line distance was measured along the line bisecting adjacent transects in progression from the first to last transect.

Hydraulic gradient was also measured using survey techniques. It represents the change in bed elevation over the distance separating adjacent transect thalwegs. The channel thalweg is the deepest point of a given cross-section of the channel (Bovee and Milhous 1978). Transects were chosen which bracketed the sampling area and were approximately 3 to 5 meters apart.

The final variable measured was the in-stream distance of the sampling areas from the lake outlet. A map measuring device was used to record the distance in kilometers of Kelsey Creek stations from Larsen Lake and the downstream distance of Bear Creek station from Paradise Lake. This variable is believed to be related to watershed area in that sediment yield generally increases in a downstream direction within a given stream. Sediment yield is expected to influence bed composition.

6.2 Gravel Sampling and Analysis, 1981

During the week of April 13, 1981, substrate samples were taken at four additional sites on each stream. Locations of these sites are described in Section 6.5.1. Samples were processed in a manner similar to that described above, but with the percent of fines less than .841 mm as the only measurement variable. Within-stream variations in the percent of fine sediment were isolated with a distribution-free test (Hollander and Wolfe 1973) and Dunn's multiple comparison procedure. The Mann-Whitney U-test was used to statistically compare the percent of fine sediment in the urban and control streams.

6.3 Streambed Stability

The extent of streambed scour and fill occurring in the study streams was determined through the combined use of scour chord and cross-sectional surveys (Miller and Leopold 1963). The scour chords consisted of heavy chain (link length 4 cm, width 2.2 cm) cut into lengths of 76 to 90 cm driven into the stream bed with a driving rod and tube. After removal of the tube, the scour chord was left embedded vertically in the substrate with several links lying on the surface. If scour occurred at the site between observations, a portion of the chain would be exposed and fixed by the current to lie flat against the redefined streambed. Fill occurring after the initial scour would leave the chain buried in this position. At approximately one-month intervals, a Wild T1-A theodolite and stadia rod were used to determine the elevation of the stream bed and of the excavated scour chain. Chains were installed approximately 1 m behind each standpipe at all the instream bioassay sites (described in Section 6.5.1). In addition, the following five sites were chosen exclusively for determination of scour.

KS1 — Kelsey, below Kelsey Creek Business Park, 13701-13715 Bel-Red. Rd.
KS2 — Kelsey, below Palisades Apts., 140th Ave. N.E. and N.E. 12th
KS3 — Kelsey, 45 m downstream of crossing of Kelsey Creek by N.E. 8th
BS1 — Bear, at foot of N.E. 165th
BS2 — at foot of N.E. 162nd

The depth of scour, averaged by site, measured in the two streams during three time intervals (Dec. 12, 1980 - March 27, 1981, March 24 - July 9, and July 9 - August 1) was statistically compared using a Mann-Whitney U-test. It was expected that the depth of scour which occurred between sampling dates would be related to the maximum discharge recorded during that time interval. To test this hypothesis, the average depth of scour at each site was regressed upon the peak instantaneous discharge as measured at U.S.G.S. gaging sites on each stream, and the significance of the relationship determined with a standard

F-test. An a posteriori test, the simultaneous test procedure (Sokal and Rohlf 1969) was used to test for differences among the slope coefficients of the regressions for sites at which a significant discharge-depth of scour relationship was demonstrated.

For an array of discharges, the percentage of streambed scoured to a given range in depth was estimated directly from the discharge-scour relationship quantified by field measurements. The percent of stream where scour depth is an interval D_i for a discharge of Q_J is given by

$$f(D_i Q_J) = \frac{\text{number of sites whose average scour is in the depth interval } D_i}{\text{number of sites on stream}}$$

The probability that an incubating egg would be swept away by a discharge of a given magnitude was then estimated by multiplying the percentage of streambed where scour in the interval D_i occurred by the probability that an egg would be deposited at that depth or less. The results compared for each depth interval were then summed to obtain the probability that an incubating egg would be scoured by a discharge of magnitude Q_J.

$$P(S_{Q_J}) = \sum_i f(D_i, Q_J)\ E(D_i)$$

$P(S_J)$ = probability egg will be scoured by discharge of Q_J

$E(D_j)$ = species-specific distribution function for depth of egg in redd.

The distribution of redd depths for coho salmon and cutthroat trout was established from a review of previous research. The depth at which coho salmon deposit their eggs has been studied by both Burner (1951) and Briggs (1953). However, the coho salmon studied by Briggs were significantly greater in length (mean - 77 cm) than those measured by Burner (mean length 61 cm). As the length of the adult coho salmon which spawn in Kelsey Creek is in the range of 55-65 cm, only the data of Burner were used to develop the distribution function for coho salmon redd depths.

The mean depth of coho salmon redds in Burner's sample was 20.3 cm, and the minimum observed depth was 7.6 cm. Under the assumptions that redd depths are normally distributed, and that only 2.5 percent of all coho salmon redds are at a depth of less than 7.6 cm, the distribution function $E(D_i)$ may be estimated as

$$E(D_i) = \theta\left(\frac{D_i - 20.3}{6.3}\right)$$

Similarly, the mean of the reported cutthroat redd depths (Cramer 1940, Kiefling 1978, Wydoski and Whitney 1979, Smith 1941) is 13.2 cm, but, unfortunately, none of these authors report the minimum depth observed. If it is assumed that only 2.5 percent of the cutthroat eggs are buried at a depth of less than 6.3 cm, then $E(D_i)$ has the form

$$E(D_i) = \theta\left(\frac{D_i - 13.2}{3.4}\right)$$

Graphs of the derived distribution functions for coho salmon and cutthroat trout redd depths are shown in Figure 15.

6.4 Intragravel Water Quality

6.4.1 Intragravel Water Quality, 1979-80

The suitability of a gravel environment as a medium for salmonid eggs and

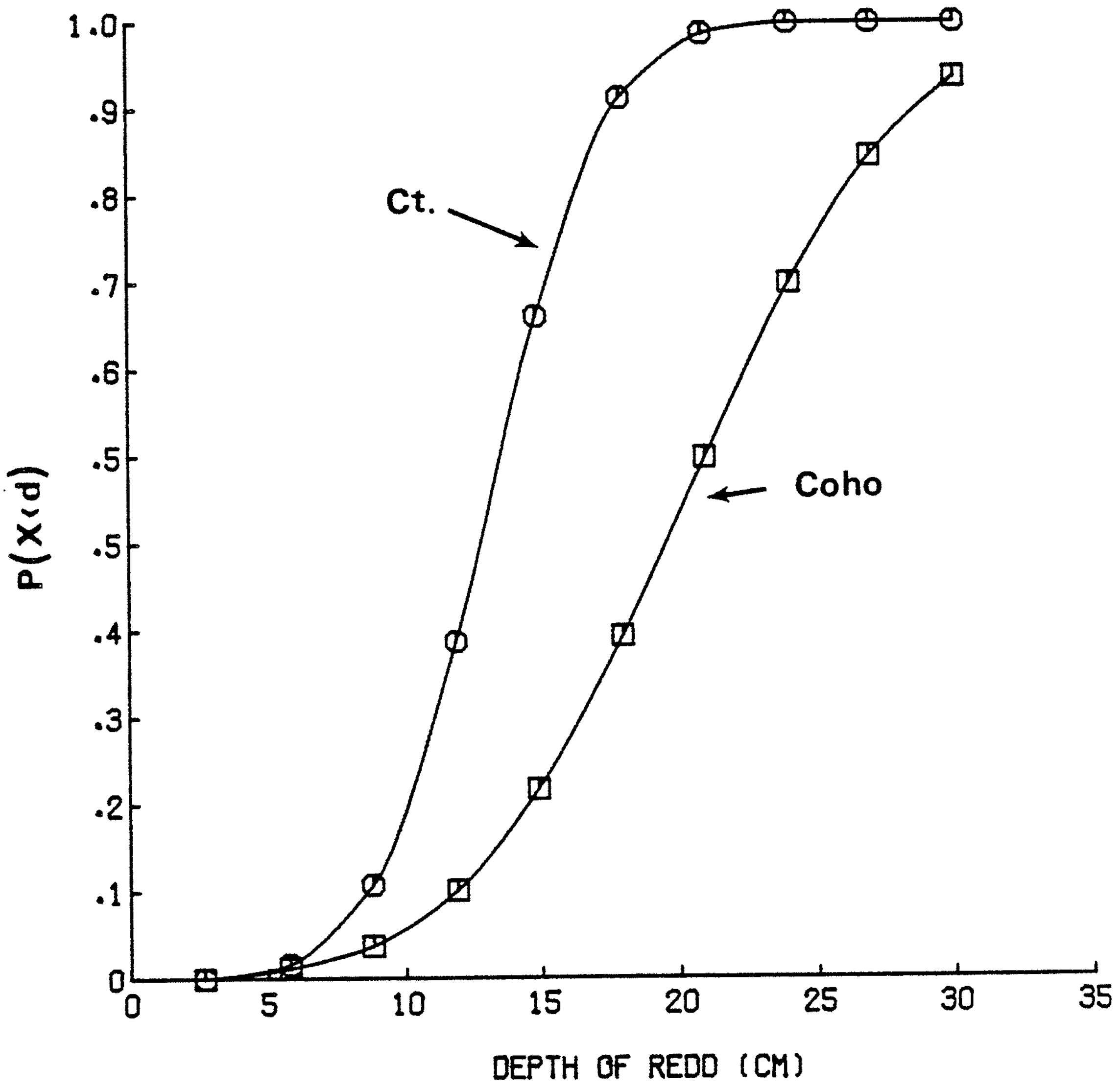


Figure 15. Derived distribution functions for the depth of cutthroat trout and coho salmon redds.

alevins is dependent on, among other factors, dissolved oxygen concentration and temperature (Ringler and Hall 1975), pH (Peterson et al. 1980), and unionized ammonia level (Rankin 1979) of intragravel water. It also depends on the state of adaptation of the fish to these environmental parameters (Schofield 1974). The present work was designed to document changes in intragravel water quality over a one-year period in Kelsey and Bear Creeks.

Three intragravel water samples were collected from each station on the ten dates indicated in Table 5. The samples were drawn from permanently installed standpipes spaced approximately 15 m apart in each station which were driven 30 cm into the stream bed. The pipes were a modified design of the Mark IV groundwater standpipe described by Terhune (1958). Standpipes were constructed of PVC plastic, and measured 3.175 cm in diameter and 90 to 120 cm in length. The lower 8 cm of each standpipe was perforated with forty-eight 3.2 mm holes divided equally among four rows. A wooden point was fixed to the bottom of the standpipes to facilitate insertion into the gravel. The tops of the standpipes remained above the water level during all but the most severe floods.

The maintenance of the standpipes required removing silt which gradually accumulated inside the pipes between sampling dates. This was done at least 24 hours prior to sampling to give the standpipes sufficient time to reach equilibrium with the surrounding water. Standpipes which were disturbed or removed by flood-borne debris during the winter months were restored to proper positions at least one day before sampling.

Water samples were collected during the early morning hours on the sampling dates since dissolved oxygen minima are normally observed in streams at sunrise. A manually operated "Rabbit pump" was used to siphon water from

Table 5. Sampling dates for intragravel water quality parameters in Kelsey and Bear Creeks.

1979	1980
August 2	January 27
September 4	March 5
October 5	April 11
November 6	May 5
December 9	June 6

the pipes into 300-ml glass DO bottles. Two samples were obtained from each pipe — one for dissolved oxygen analysis and the other for determination of pH and ammonia concentration. Intragravel water temperatures were recorded to the nearest 0.1°C by lowering a long-stem laboratory thermometer into the pipes.

Dissolved oxygen samples were fixed immediately after collection with the conventional reagents used in the azide modification of the Winkler method (American Public Health Associated et al. 1975). Following treatment with 2 ml each of manganese sulfate solution, alkali-iodide reagent, and concentrated H_2SO_4, the samples were returned to the laboratory and titrated with 0.0250 N thiosulfate solution. Dissolved oxygen concentrations were recorded in mg/l.

The water samples obtained for ammonia and pH determination were analyzed within 3 hours of collection. A pH meter equipped with a glass electrode was used to measure pH to the nearest 0.1 pH units.

The method used to determine total and un-ionized ammonia concentrations involved alkalization of the samples to pH 12 and subsequent measurement of total ammonia with a gas-permeable membrane electrode (ORION model 9S-10). A calibration curve was prepared for each series of samples by recording the electrode potentials of standard ammonia solutions (0.1, 0.5, and 1.0 mg total ammonia $\cdot$ l^{-1}). The amount of un-ionized ammonia present in the samples, knowing the sample pH and temperature, was calculated using the following formulae (Emerson et al. 1975):

$$pK_a = 0.09018 + (2729.92/T^{-1})$$

where T is the temperature in °K and

$$f = 1 / (10^{pK_a - pH} + 1)$$

where f is the fraction of total ammonia present as NH_3-N.

The reliability of the ammonia probe decreases at low ammonia concentrations and is unsuitable for concentrations below 0.1 mg $NH_3\text{-N} \cdot l^{-1}$ (Barka 1973). Our results from the first three sampling dates indicated that a more sensitive means of measuring NH_3-N was necessary and subsequent determinations were made using the conventional spectrophotometric phenolhypochlorite method (Solorzano 1969).

Water quality data was subjected to analysis of variance and multiple comparison procedures. Specific objectives included determining spatial and temporal patterns in the parameters, differences between the two streams, and the relationship between bed composition and dissolved oxygen levels. The latter objective involved a regression of dissolved oxygen concentrations against mean percent fines and d_g values. The dissolved oxygen samples were collected from standpipes positioned within the gravel sampling areas.

6.4.2 Intragravel Water Quality, 1980-81

In conjunction with the instream embryo bioassays, intragravel water quality parameters were monitored throughout the winter and spring of 1980-81. At least one month prior to commencement of the instream bioassays, standpipes (Wickett 1954) were placed at each site to facilitate the removal of interstitial water samples. The features of several standpipe models were reviewed (Wickett 1954, Terhune 1958, Gangmark and Bakkala 1958, Coble 1961) and modified to increase their durability and resistance to scour. The final design consisted of a 70-90 cm length of 1/2", schedule 80 PVC pipe, plugged at one end with a rubber stopper. In the interval located between 23 and 33 cm from the opposite end, 36 3.2 mm holes were drilled. Each standpipe was embedded in the streambed by allowing it to slide through a hollow steel tube previously forced through the sediment by means of a driving rod and a 11.3 kg sledge. After removal of the steel tube, the standpipe was capped and adjusted so that approximately 5 cm protruded above the stream bed, thus assuring that the drilled holes were at the depth at which salmonid eggs are generally buried (18 to 28 cm) (Burner 1951). At each site a row of five standpipes, arranged perpendicular to stream flow and separated by 33-50 cm, was established.

Interstitial water samples were obtained from the standpipes by removing the cap and attaching an extension tube by means of a compression fitting. Before removing the water sample, the standpipe was siphoned for 60 seconds (or until dry) to remove any surface water which might have entered during installation of the extension tube. Siphoning was accomplished by means of a small hand-operated "Rabbit pump." Interstitial and surface water was sampled for dissolved oxygen immediately after sunrise at 3 to 20-day intervals during the instream bioassay test periods. Samples were analyzed using the azide modification of the Winkler method (American Public Health Association et al. 1975). A distribution-free test (Hollander and Wolfe 1973) and Dunn's multiple comparison procedure were used to test for and isolate significant differences in the dissolved oxygen concentration at sites within a stream. Data from each stream were compared with a Mann-Whitney U-test.

Interstitial water was sampled for dissolved lead on one occasion. Samples were removed using the process described above and placed in acid-rinsed bottles. Analysis was performed by J. Richey (U.W., College of Engineering), using flameless atomic absorption and standard procedures (A.P.H.A. 1975).

6.5 Embryo Bioassays

Two experiments were devised to isolate the effects of the three variables (scour, sedimentation-dissolved oxygen, toxicants) hypothesized to adversely impact pre-emergent salmonids in urban streams. The first experiment, which included monitoring of the gravel environment and instream embryo bioassays, was designed primarily to answer questions relating to streambed scour and sedimentation. The objective of the second, conducted concomitantly in a streamside incubation box, was to assess the additional embryo mortality induced by toxicants.

6.5.1 Instream Embryo Bioassays

Instream embryo bioassay site selection was based on three criteria: the site must be representative of salmonid spawning areas within the stream, accessible to sampling gear, and sufficiently large and homogeneous to allow within-site replication. Site locations and designations are as follows:

KE1 — Kelsey, 90 m upstream of the confluence of Kelsey and Valley Creeks
KE2 — Kelsey, within Sandpiper East Apt. complex, 13638 N.E. 13th St.
KE3 — Kelsey, 45 m downstream of crossing of Kelsey Creek by 134th Ave. N.E.
KE4 — Kelsey, Kelsey Creek Park
BE1 — Bear, 18 m below crossing of Bear Creek by the Woodinville-Duvall Rd.
BE2 — Bear, at foot of N.E. 167th Pl.
BE3 — Bear, at foot of N.E. 162nd St.

BE4 — Bear, at foot of 200th Ave. N.E.

A variety of techniques was considered during the development of the instream bioassay methodology. Sampling natural redds hydraulically (McNeil 1962) was precluded due to the limited escapement, particularly of coho, to Kelsey Creek. Emergent fry traps (Phillips and Koski 1969) were judged unlikely to survive periods of extensive scour. In order to avoid damaging natural redds, many researchers (Gangmark and Bakkala 1960, Coble 1961, Phillips and Campbell 1962, Turnpenny and Williams 1980) have placed eggs in some type of bag or enclosure, buried it, and estimated embryo survival from the ratio of dead to live eggs in the enclosure upon recovery. However, recent studies have cast doubt as to how well this method approximates intragravel conditions. Harshbarger and Porter (1979) found that "boxes impeded water movement and induced sediment deposition in and immediately around the egg box" and that "eggs confined in boxes are more susceptible to fungal agents than eggs in intragravel plants."

To circumvent these problems, the newly developed salmon egg-planting device (SEPD) (Jones et al. 1977, White 1980a, b) was used to plant salmonid eggs into the stream bed. The SEPD is basically a hollow PVC tube (2.5 cm in diameter, 100 cm long) which is driven 20-25 cm into the stream bed with the assistance of water pressure supplied by a centrifugal pump. Water flowing from the probe end also acts to flush out intragravel fines and develop water flow channels. After insertion of the SEPD, one valve is used to halt the flow of water, while a second valve at the exposed end of the device is opened to allow planting of the artificial redd. The eggs are then placed into the tube, allowed to settle for 10-15 seconds, and the SEPD removed. Each artificial redd was constructed 4-10 cm directly upstream of the standpipe.

Two instream bioassays were conducted. The first utilized coho eggs obtained on Dec. 1 and 3, 1980, from the hatchery operated by the University of Washington School of Fisheries. After allowing the eggs to water-harden for two hours, groups of 300 were counted out and placed in one-quart jars filled with water. Jars were transported to the stream in an ice chest and allowed to acclimate to stream temperature. Planting of the 20 artificial redds at each stream was accomplished within five hours. For the second bioassay, early eyed-stage rainbow eggs, also supplied by the School of Fisheries, were used. Groups of 200 eggs were counted, placed with water in a Zip-Loc Bag, and transported to the planting site immersed in a water-filled ice chest. After acclimation of the eggs, artificial redds were constructed at the first, third, and fifth standpipes at each site. Planting of eggs for the second bioassay was conducted on April 23, 1981. Due to an uncooperative landowner, it became necessary to relocate BE4 at scour site BS1. This site was thereafter designated BE4B.

Embryonic survival at each site was determined at 3-6 week intervals during the initial stream bioassay and at approximately 12-day intervals during the second. Each redd was sampled hydraulically (McNeil 1962) until further probing yielded no results. The eggs and alevins recovered were preserved with Stockard's solution and 7 percent formalin, respectively, and taken to the laboratory for counting. Survival was estimated from a modified version of McNeil's (1966) equation 4:

$$S_{iJ} = \frac{L_{iJ}}{c_i N}$$

where

S_{iJ} = survival at ith site on Jth sampling date

L_{iJ} = live embryos recovered at ith site on Jth sampling date

N = number of eggs planted

c_i = correction term for the efficiency of sampling at each site.

c was estimated using the relationship

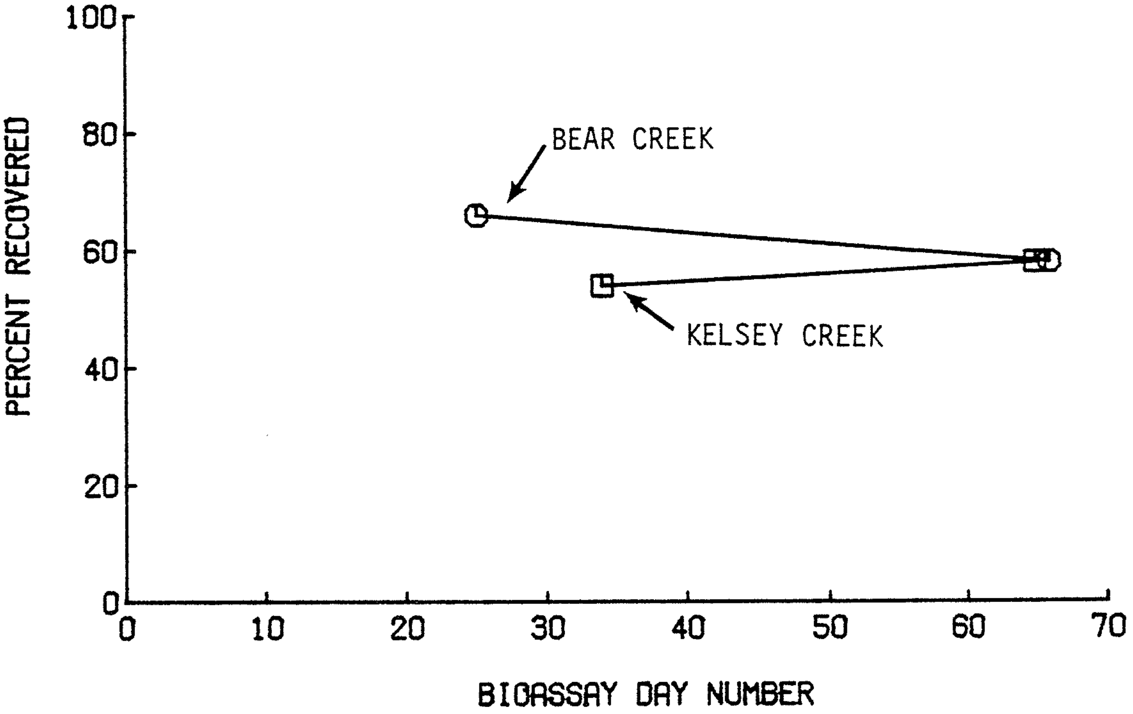
$$C_{iJ} = \frac{L_{iJ} + D_{iJ}}{N}$$

where

D_{iJ} = number of dead embryos recovered at ith site on Jth sampling date.

This method works well if no embryos (dead or alive) are removed from the artificial redd during the bioassay period. Such was the case during the winter bioassays (Fig. 16), when the water temperature was too cold for rapid decomposition to occur and scour did not approach the depth of the eggs. In neither stream did the median number of eggs recovered differ significantly between the first and second sampling period.

During the spring bioassays this was not the case. In both streams, as time progressed, a decline in the total number of eggs recovered was evident (Fig. 16). In particular the median number of eggs recovered from Kelsey Creek during the third sampling date is significantly less ($\alpha \leq .028$) than the number recovered during the initial sampling period. If it is assumed that this loss of embryos was due to the decomposition of dead eggs, as seems possible, then the method previously outlined will overestimate survival. For this reason, for the second and third sampling dates, a mean efficiency for each stream was calculated from the C_{iL}. That is, the efficiency was estimated as



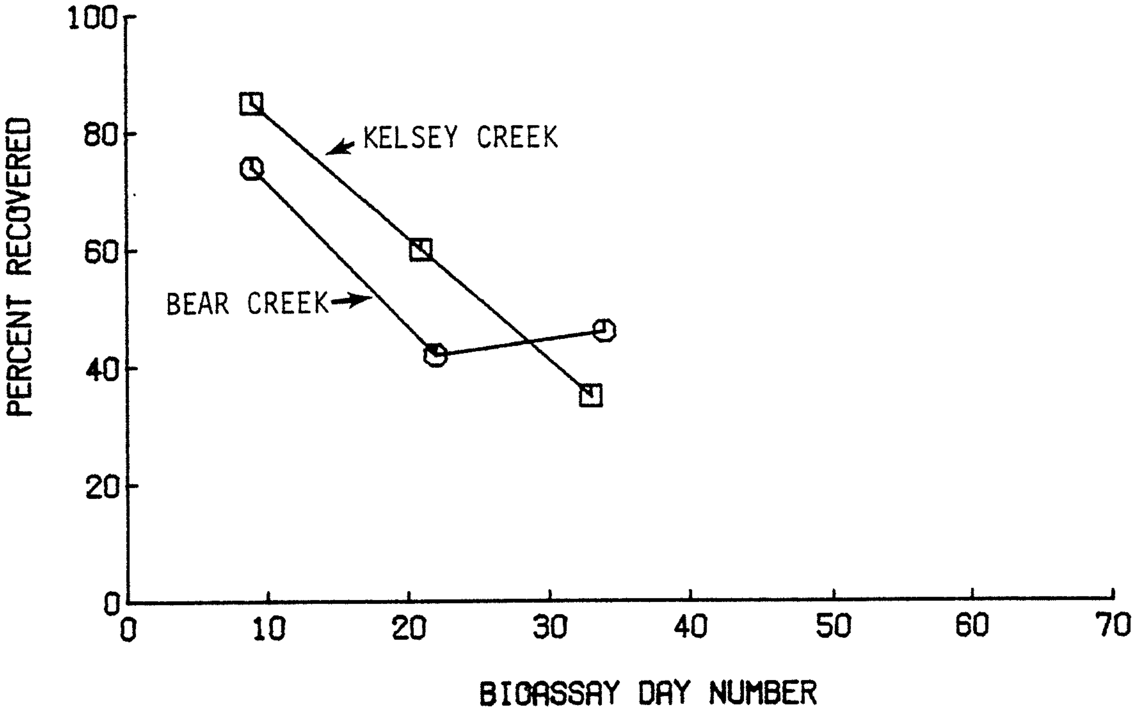


Figure 16. Percentage of embryos planted that were recovered at each sampling date during the winter (upper) and spring (lower) instream embryo bioassays.

$$\bar{C}_k = \frac{1}{s_k} \sum_{i}^{s_k} C_{ilk}$$

where the additional subscript k denotes stream, and s is the number of sites initially sampled on that stream.

A distribution-free test for the slope coefficient (Hollander and Wolfe 1972) was employed to statistically determine if a significant relationship existed between embryo survival and the mean dissolved oxygen concentration to which it was exposed. Survival in each stream was compared by means of a Mann-Whitney U-test.

6.5.2 Streamside Embryo Bioassays

The streamside bioassay was designed as a simple, first approximation to the detection of water quality problems, exclusive of sedimentation, which might be hampering the survival of alevins. Bioassays were conducted in an embryo incubation box established jointly by FRI, Seattle METRO, the City of Bellevue, and the Washington State Department of Fisheries. The box was located near the mouth of Kelsey Creek at Kelsey Creek Park (Figure 6) where toxicants, if present within the stream system, would be most likely to be encountered.

Briefly, the incubation box consisted of a plywood enclosure through which natural stream water was gravity-fed at a rate of 37-58 l/min. Sediment was removed from the inflow by means of two settling boxes. Readers desiring a more extensive description of the embryo incubation facilities are referred to Bauersfeld et al. (1981).

Sockeye eggs for the bioassay were acquired from an incubation box located on Valley Creek, which had been stocked by WDF with eggs originally collected from an artificial spawning program on the Cedar River. Approximately 1-2 days prior to hatching (January 26, 1981) 300 eggs were transported in a water-filled ice chest from Valley Creek to the Kelsey Creek streamside incubation bioassay site. Vibert boxes (Harris 1973) lined with mylar were filled with biorings and 20 sockeye eggs. At 7-10 day intervals thereafter, a minimum of two Vibert boxes was removed and mortalities within them noted. The percent survival in each box was estimated as

$$S = \left[1 - \frac{M}{20} \right] \cdot 100$$

where

M = number of mortalities found.

Dissolved oxygen and lead concentrations in water samples from the streamside incubation box were determined coincidentally with the routine instream sampling program.

6.6 Population Dynamics

6.6.1 Study Site Selection

If estimates of salmonid abundance and production are to be extrapolated to an entire stream system, the assumption must be made that the study areas sampled are representative of the rest of the stream. The random selection of a large number of study areas would satisfy this assumption; however, both time constraints and site accessibility precluded this approach to sampling. Our method of site selection involved dividing equal lengths of Kelsey Creek and Bear Creek into subsections from which study areas were systematically chosen on the basis of their physical characteristics, accessibility and relative position along the length of the stream. In Kelsey Creek 5 stations approximately equal in length (45-55 m) and distance from one another were selected along the length of the stream where salmonids resided and access permitted sampling. Care was taken to include pool and riffle habitat in study areas ranging from moderate to severe channel alteration. The abundance of fish at these stations (KC1-KC5) was estimated on 13 occasions during the period April 1979 to August 1981. Five additional study areas (KC6-KC10), established adjacent to the regular stations, were sampled in August and December 1979, to avoid repeatedly stressing fish in the latter stations. Three study areas in Bear Creek (BC1-BC3) were chosen using the same criteria applied to Kelsey Creek. Disturbed areas in Bear Creek are uncommon and were therefore not included as sampling sites. The inaccuracies in population estimates and production statistics which accrue to non-random sampling procedures are not believed to be large and should be consistent from one sampling period to the next.

In addition, the fish populations of representative sections of major Kelsey Creek tributaries were assessed once in late August 1981. Designations and locations of these study sites are as follows (Fig. 6):

VC1 — Valley Creek, extending upstream from the Innis Glen Apartments on 140th N.E.

WC1 — West Tributary, extending downstream from approximately 50 m below the N.E. 8th bridge

WC2 — West Tributary, extending downstream from the N.E. 1st bridge

RC1 — Richards Creek, extending downstream from the Richards Road bridge

6.6.2 Water Surface Area Estimates

Surveying techniques used to measure the total wetted area of each station during low flow periods required a Wild II theodolite, stadia rod, and a 100-meter surveyor's tape. During a field survey 12 to 15 permanent transects were established at each station to enable cross-sectional measurements of water depth and velocity. Transect headstakes were positioned primarily to measure a discrete biotope: pools, runs, riffles, etc. They were also placed at hydraulic controls, defined as physical features, either natural or man-made, which indicate a stage-discharge relationship (Bovee and Milhous 1978).

Starting at the downstream end of a station, the theodolite was mounted on a tripod in the middle of the stream, leveled, and a reference reading taken on a permanent benchmark. For each transect the horizontal angles from the instrument to the headstakes and to the water's edge on each side of the stream were measured relative to the azimuth connecting the instrument and the benchmark. Horizontal angles were also measured to prominent physical features along the margin of the stream, such as large rocks, tree trunks, and other outcroppings. Distance measurements included stream width at each transect, the distance along thalwegs (the longitudinal line connecting points of maximum water depth along adjacent transects), and the distance between the theodolite and locations at which angular measurements were taken. It was necessary to reposition the theodolite once or twice as required to make sights on upstream transects; this was accomplished by measuring all of the possible angles and distances between the benchmark and instrument positions. Horizontal angles were recorded to the nearest 5 seconds and distance to the nearest centimeter.

Reference maps of the surface area at each study site were constructed from surveying data on 1 mm graph paper. The boundary of the stream, defined primarily by transect locations, was referenced by a co-ordinate system and coded for computer analysis. The Synagraphic Mapping program (SYMAP) (Dougenik and Shechan 1975) was used to obtain areal estimates of the space enclosed by the boundary outline. These estimates were verified by counting the number of grids of stream surface area on the reference map.

All sites on Kelsey and Bear Creek were resurveyed in August 1981 to allow for channel modification. After that date, only BC2, KC3, and KC4 were judged to have undergone significant alteration. These sites, as well as those on the Kelsey Creek tributary streams, were once again surveyed in August 1981.

The wetted surface area at each of the stations KC6 through KC10 was determined by multiplying the midline length of the sample reach by the average stream width, measured perpendicular to the midline at 6 locations within the study area.

6.6.3 Fish Sampling

The collection of salmonid fishes in the study areas was accomplished by blocking the upper and lower end of each section with 0.5 mm mesh nets and making three to eight successive passes with a Smith-Root electroshocker. Following each pass the salmonids were separated by species, anaesthetized with tricaine methane-sulphonate (MS 222), and individual lengths and weights recorded. Length was measured to the nearest mm and weight to the nearest gram until April 1980, after which weight was determined to the nearest 0.10 gram. Scale samples were obtained from each general length frequency size class in order to verify age determinations. The sex of older cutthroat trout was ascertained when possible. Beginning in July 1980, all salmonids of length greater than 55 mm were freeze-branded (Everest and Edmundson 1967) with a tool which had been cooled in a slurry of dry ice and acetone. The date and site-specific brands used are recorded in Table 6. Many young-of-the-year fish captured in the spring of 1981 were too small to allow freeze branding. A portion of these fish was marked by excision of the adipose fin. With the exception of a single sampling date in August 1981, detailed data on the number and weight of non-salmonids were not recorded.

Two simple measurements of fish health were instituted commencing in December 1981. Each fish was examined for evidence of fin erosion or anomalies in respiratory apparatus, and the conditions recorded as either present or absent. Three specimens which displayed damaged gills in August 1981 were transported live to the laboratory of D. Elliott (School of Fisheries, University of Washington) for intensive examination. Gills from these fish were preserved in Bouin's fixative, embedded in paraffin, sectioned, stained with hematoxoylin-eosin or Giemsa according to standard procedures, and examined by light microscopy.

6.6.4 Estimation Procedures

The population size of each species-age group was estimated by catch-effort and, where possible, mark-recapture methods. Different catch-effort formulae were employed depending upon the number of passes made. For two or three pass estimates, exact solutions to the general model are possible. These are detailed below.

Two-pass method (Seber and LeCren 1967):

$$\hat{N}_E = \frac{n_1^2}{(n_1-n_2)}$$

Table 6. Site and date specific brands employed during the marking program.

	Site							
Date	KC-1	KC-2	KC-3	KC-4	KC-5	BC-1	BC-2	BC-3
7/80	ALPU[1]	ALPD	ALPB	ALPF	ALAU	ALPU	ALPD	ALPF
9/80	ALAD	ALAF	ALAB	ARAU	ARAD	ALAD	ARPF	ARPB
11/80	ARAF	ARAB	ARPU	ARPD	ARPF	ARAB	ARAD	ARAU
12/80	ALPU	ALPD	ALPB	ALPF	ALAU	ARPU	ALAU	ALAF
3/81[2]	BLPU	BLPD	BLPB	BLPF	BLAU	BLPB	BLPD	BLPU
5/81	BRPU	BRPF	BLAB	BRPD	BRPU	BLAB	BLAD	BLAB
7/81	BRAF	BRAB	BRAU	BRAD	BRPF	BRAB	BRAD	BRAU

[1] Character denotation:

Character 1: Position with respect to the lateral line.
A — Above
B — Below

Character 2: Side of fish.
L — Left
R — Right

Character 3: Lateral position.
A — Anterior
B — Posterior

Character 4: Symbol orientation.
F — Forward
B — Backward
U — Up
D — Down

[2] Adipose fin excised as well.

and

$$V(N_E) = \frac{n_1^2 \; n_2^2 \; (n_1 + n_2)}{(n_1 - n_2)^4} \quad ,$$

where

n_1 = catch on first pass

n_2 = catch on second pass

Three-pass method (Junge and Libosvárský 1965):

$$\hat{N}_E = \frac{6x^2 - 3xy - y^2 + y(y^2 + 6xy - 3x^2)^{\frac{1}{2}}}{18(x - y)} \quad ,$$

$$\hat{p} = \frac{3x - y - (y^2 + 6xy - 3x^2)^{\frac{1}{2}}}{2x} \quad ,$$

$$\hat{q} = 1 - \hat{p} \quad ,$$

and

$$V(\hat{N}_E) = \frac{\hat{N}(1 - \hat{q}^s) \; \hat{q}^s}{(1 - \hat{q}^s)^2 - (\hat{p}s)^2 \; \hat{q}^{s-1}} \quad ,$$

with

$\hat{p}$ = the probability of capturing a randomly chosen fish in a given pass,

s = the number of passes,

x = $2n_1 + n_2$

y = $n_1 + n_2 + n_3$

If more than three passes are made, the following equation (Zippin 1956) must be solved iteratively to obtain an estimate for N:

$$\frac{\hat{N}_2 - y}{\hat{N}_E} = s\left[\frac{s\hat{N}_E - x - y}{s\hat{N}_E - x}\right]$$

and p may then be estimated by

$$\hat{p} = \frac{y}{sN_E - x} \quad .$$

The variance of N is estimated in the same manner as described for the three-pass methodology.

All of the catch and effort methods discussed above require that three assumptions be met (Seber 1973).

1. The population is closed during the sampling period.
2. In a given pass, the probability of capture is equal for each fish exposed to capture.
3. The probability of capture p remains constant from pass to pass.

An independent estimate of population size was made from the mark-recapture data in conjunction with the Jolly-Seber model (Jolly 1965, Seber 1965). The necessary formulae are

$$\hat{M}_i = \frac{(Q_i + 1)\ Z_i}{(r_i + 1)} + m_1 + 1 \quad , \qquad (i = 2, 3, \ldots, d - 1)$$

$$\hat{N}_{MR,i} = \frac{\hat{M}_i\ (n_i + 1)}{(m_i + 1)} \quad , \qquad (i = 2, 3, \ldots, d - 1)$$

where

$\hat{N}_{MR,i}$ = estimated total number in the population just before the i^{th} sample date

$\hat{M}_i$ = estimated total number of marked fish in the population just before the i^{th} sample date

n_i = number caught on the i^{th} sample date

m_i = number of marked animals caught on the i^{th} sample date

Q_i = number of marked fish released after the i^{th} sample date

r_i = number of marked animals released after the i^{th} sample date which are subsequently recaptured

d = number of sample dates.

The variance of $\hat{N}_{MR,i}$ may be estimated by

$$V(\hat{N}_{MR,i}) = \hat{N}_{MR,i}\,(\hat{N}_{MR,i} - n_i)\left[\frac{\hat{M}_i - m_i + Q_i}{\hat{M}_i}\left(\frac{1}{r_i} - \frac{1}{Q_i}\right) + \left(1 - \frac{m_i}{n_i}\right)\left(\frac{1}{m_i}\right)\right]$$

$$+ \hat{N}_{MR,i} - \sum_{h=0}^{i-1} \frac{\hat{N}^2_{MR,i}(h)}{\hat{I}_h} \qquad (i = 2, 3, \ldots, d - 1)$$

where

$\hat{I}_h$ = number of new fish joining the population in the interval from the h^{th} to the $(h + 1)^{th}$ sampling date which are still alive and in the population at the $(h + 1)^{th}$ sampling date,

$\hat{N}_{MR,i}(h)$ = number in the population on the i^{th} sampling date which first joined the population between the h and $(h + i)^{th}$ sampling $(1 \leq h \leq i - 1)$.

Seber (1973) states that the final two terms in the expression above, the latter of which is difficult to compute, are generally small and can be ignored in most cases. This simplified expression for the variance of $\hat{N}$ was used for all computations.

It is essential that all fish captured retain their marks throughout the experimental period if accurate population estimates are to be made. For this reason the 13-month period in which marking was conducted was divided into segments consisting of four sampling dates apiece (duration of approximately six months). Thus, beginning in July 1980, segment 1 consisted of sample dates in July, September, November, and December; segment 2 included September, November, December, and April; and, moving iteratively forward, the final segment consisted of April, May, June, and August, 1981. As the Jolly-Seber model permits estimation of the population size at all but the first and last sampling dates of each segment, it may be seen that two estimates of N_i result from this iterative technique. Blower et al. (1981) have noted that the last estimate of N_i obtained in a Jolly-Seber experiment is subject to sampling error; therefore, it was deemed the most likely to be inaccurate and was not utilized in further analysis if an alternative estimate from the mark-recapture data was possible.

The estimates of population size obtained from the catch-effort and mark-recapture methodologies were pooled by weighing each by its associated variance:

$$\hat{N}_p = \frac{\dfrac{\hat{N}_E}{V(\hat{N}_E)} + \dfrac{\hat{N}_{MR}}{V(\hat{N}_{MR})}}{\dfrac{1}{V(\hat{N}_E)} + \dfrac{1}{V(\hat{N}_{MR})}}$$

which has an estimated variance of

$$V(\hat{N}_p) = \frac{1}{\frac{1}{V(\hat{N}_E)} + \frac{1}{V(\hat{N}_{MR})}}$$

The Jolly-Seber model also allows estimation of the survival (θ_i) of fish between sampling periods:

$$\hat{\theta}_i = \frac{\hat{M}_{i+1}}{\hat{M}_i - m_i + Q_i} \quad , \quad (i = 2, 3, \ldots, d-2)$$

with a variance of

$$V(\hat{\theta}_i) = \hat{\theta}_i^{\,2} \left[\frac{(\hat{M}_{i+1} - m_{i+1})(\hat{M}_{i+1} - m_{i+1} + Q_{i+1})}{\hat{M}_{i+1}^2} \left(\frac{1}{r_{i+1}} - \frac{1}{Q_{i+1}}\right) + \left(\frac{\hat{M}_i - m_i}{\hat{M}_i - m_i + Q_i}\right)\left(\frac{1}{r_1} - \frac{1}{Q_i}\right) + \frac{1 - \hat{\theta}_i}{\hat{M}_{i+1}} \right] , \quad (i = 1, 2, \ldots, d-2)$$

As was previously discussed with respect to the N_i, two estimates of θ_i result from the iteration technique employed. Where possible, only the latter estimate for survival obtained from the analysis of a given segment was employed. More precisely, if θ_{iJ} is the survival of fish between sample date i and i-1, and J is a subscript denoting the time segment, then θ_{21} was preferred to θ_{12}.

The rate at which fish disappeared from the sample site was estimated by

$$\hat{L}_i = \frac{1 - \theta}{\Delta t}$$

where Δt is the number of days between sample dates.

The number of immigrants (or births) between sample dates i and i+1 which survive to i+1 may be estimated as

$$\hat{I}_i = \hat{N}_{p,i+1} - \hat{\theta}_i (\hat{N}_{p,i} - n_i + Q_i) \quad , \quad (i = 2, 3, \ldots, d - 2)$$

Although the variance of $\hat{I}_i$ is difficult to obtain due to correlations among the parameters involved, it will not be greater than one estimated solely from the catch-effort data (D. Chapman, personal communication). Thus

$$V(\hat{I}_i) \simeq V(\hat{N}_{E,i+1}) + \hat{N}_{E,i} V(\theta_i) + \theta_i^2 V(\hat{N}_{E,i}) \quad , \quad (i = 2, 3, \ldots, d - 2)$$

The rate of replacement of fish within a study area was estimated as

$$\hat{R}_i = \frac{\hat{I}_i}{(\hat{N}_{p,i+1})(\Delta t)}$$

A length-weight relationship for both species of salmonids in Kelsey and Bear Creeks was calculated from data collected during electrofishing sessions. Using the logarithmic transform of observed lengths and weights, regression lines were calculated by the method of least squares. Statistical comparisons of the regression lines were performed in order to distinguish differences in length-weight relationships between sympatric and allopatric salmonid species. The appropriate regression equation was used to estimate the weight of fish weighing less than 1.0 gram collected on sampling dates other than July 1980.

A slightly different approach was employed to estimate fish weight from lengths after April 1980. For each species and stream, the length-weight data from each sampling date were used to define a regression equation. Statistical tests of the slope ($\hat{B}$) and elevation ($\hat{\alpha}$) coefficients were then employed to determine for which sampling dates the data might be pooled. The procedure, as detailed in Zar (p. 228-234; 1974), may be broken into the following steps:

1. Test the null hypothesis H_0: $B_1 = B_2 = \ldots B_K$;
2. If the test statistic is significant ($\alpha \leq .05$), isolate homogeneous groups with a Neuman-Keuls multiple range test;
3. For each homogeneous subset isolated in 2), test the null hypothesis H_0: $\alpha_1 = \alpha_2, = \ldots \alpha_K$;
4. For subsets in which the test statistic computed in 3) is significant, employ the Neuman-Keuls multiple range test to isolate homogeneous subsets once more.

The biomass of fish supported by each stream per m^2 of summer low-flow surface area was estimated by multiplying the mean weight of each species-age group by its abundance and dividing this by the low-flow surface area of the stream.

$$\hat{B}_i = \frac{\hat{N}_i \overline{W}_i}{A}$$

where

$\hat{B}_i$ = the estimated biomass of a given species-age group per m^2 of low-flow stream surface area on the i^{th} sampling date

A = total low-flow stream surface area sampled on the i^{th} sampling date.

Instantaneous population growth rates for weight were calculated as

$$g_W = \frac{\ln \overline{W}_{i+1} - \ln \overline{W}_i}{\Delta t} ,$$

where

g_W = instantaneous rate of growth in weight for a given species-age group;

$\overline{W}_i$ = mean weight of a given species-age group on sampling date i (includes both fish whose weight was measured directly, and those whose weight was estimated from their measured length);

Δt = number of days between sample dates i and i + 1.

Production between sampling dates was estimated as

$$P_i = g\overline{B}\,\Delta t \quad ,$$

$$\overline{B} = \frac{\hat{B}_i + \hat{B}_{i+1}}{2} \quad ,$$

where

P_i = the net elaboration of tissue between sample dates i and i + 1.

A second approach to calculating annual production, frequently used when only one sample is collected, was employed for comparison with the results of the method described above for the period of study prior to July 1980. Rather than sum production over a series of time intervals, only the density and mean individual weight of a given age group on the first (July 1979) and last (July 1980) sampling dates were used to calculate the annual production. It was thought that a comparison of the two methods and an estimation of the degree and direction of bias, if any, could be instructive.

Chi-square tests were employed to determine if a statistically significant difference existed in the proportion of fish afflicted with respiratory anomalies at each site. Sites on each stream were first tested for heterogeneity and homogeneous subsets isolated. Chi-square analysis was then applied once more to test for differences between the subsets of each stream.

6.6.5 Population Stability: Age and Year Class Comparisons

As a means of analyzing age structure stability in the salmonid populations of the two streams, correlation coefficients were used to determine the relationship between the abundance of age 0, 1, and 2 fish in July 1979 and the same age groups in July 1980. Density ($No \cdot m^{-2}$) and biomass ($g \cdot m^{-2}$) were selected as measures of age group abundance. Additional comparisons were made between April samples collected from Kelsey Creek in both years.

In a stable self-maintaining fish population the number of fish in each cohort can be expected to decrease over time. If this decline is uniform for all segments of the population the abundance of fish in a given year class at time $t + 1$ should be a function of the number of fish present at time t. Comparisons of year class density and biomass estimates between various sampling dates were made by regression analysis to determine the extent to which the size of the previous generation influences the size of the next in both streams. To the extent that regular variations in year class abundance occur, certain processes of regulation may be implicated, primarily those intrinsic to the population. The lack of a significant correlation suggests, but certainly does not confirm, regulatory mechanisms extrinsic to the population. Variations between study areas and in the level of sampling effort can be expected to contribute to the lack of fit. It should be stressed that correlation techniques do not distinguish between the effects of compensatory and extrapensatory factors on survival since correlation does not imply causation (Eberhardt 1970). Correlation analysis can, however, give some idea of the conformity of population changes between study areas, at the same time indicating the direction and degree of the observed response. It does not, however, provide a predictive equation for the fluctuations in the abundance of the selected year class. In using this approach it was assumed that the relationship between the variables is linear, that the variables are normally distributed, and that the abundance estimates are random samples from the entire population. The quantity $t = r\ (N - 2/1 - r^2)^{\frac{1}{2}}$ was compared with the Student t-distribution with N-2 degrees of freedom to test the significance of r.

6.6.6 Kelsey Creek Emigrants

The downstream netting operations were conducted at the 112th Ave. S.E.-Kelsey Creek culvert (Fig. 6), a location which was ideal in several respects. Most importantly, no stream habitat existed below this point; thus, the population sampled by the net consisted of all fish migrating from the Kelsey Creek watershed. In addition, the cement and wood structures which comprised the culvert and associated fish ladder (Fig. 17) provided an extensive framework by which the net could be firmly secured.

Netting commenced on August 3, 1980, and proceeded on a regular schedule until July 28 of the following year. Initially, the net was fished at 7 to 14-day intervals, generally during the night, but with occasional sampling during the day. The net was continuously in operation from March 31 to April 30 and was fished 2 to 7 days per week from May 4 until the end of July 1981.

Two nets were employed during the sampling program, both of which were constructed of 3-mm knotless nylon mesh mounted on a frame of 15.8-mm steel reinforcing rod. The smaller net measured 100 cm wide by 61 cm high at the

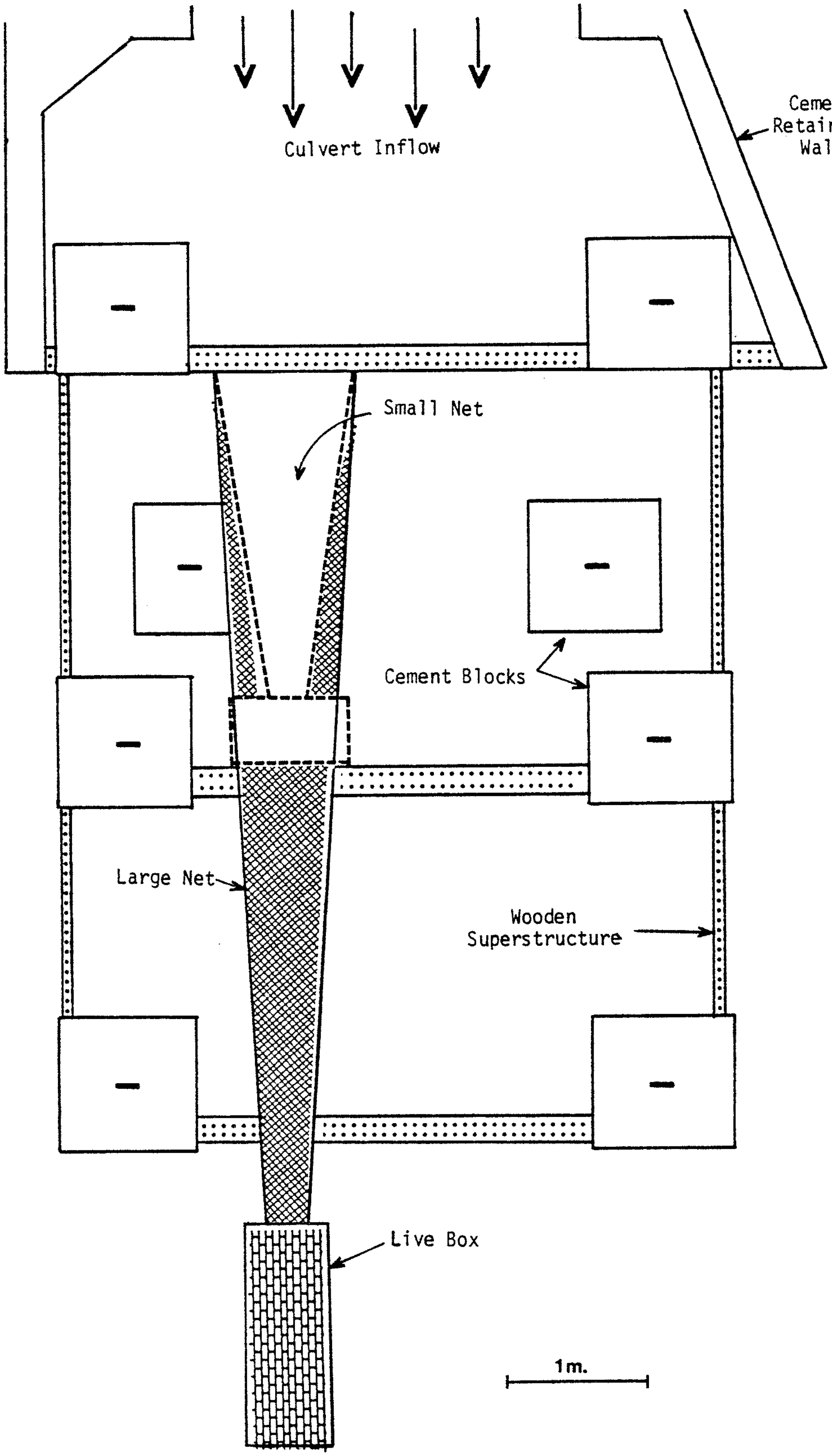


Figure 17. Schematic diagram of the small and large downstream nets.

mouth, and tapered down in a distance of 244 cm to a diameter of 15 cm. A live-box measuring 76 cm long, 46 cm wide, and 16 cm deep, constructed of wood and nylon mesh, was clamped to the cod end of the net. The larger net had a mouth 1.2 x 1.2 m, from which it tapered down in a distance of 6.1 m to a diameter of 15 cm. The live-box of the larger net measured 56 x 56 x 152 cm long.

The bottom edge of the steel frames on which the nets were mounted was secured to a wooden step in the fish ladder with 10 mm eye hooks, permanently bolted in place. The top end of the frame was held in place by two cables which ran to retaining rods mounted on the fish ladder (small net) or on shore (large net). The nets could be installed by simply sliding the frame into the eye hooks and attaching the support cables. Removal of the nets was achieved by reversing this process.

Flow conditions within the stream determined which net was used. During low-flow periods the small net was preferred because of the ease with which it could be handled. The large net was used in the winter and spring months, when the increased capacity provided by its greater height and length was required. Note that the volume of water intercepted by the small and large nets would be equal until the stream reached a stage at which overtopping of the small net occurred. Neither the large nor small net was overtopped during this study.

The efficiency of the large net was tested on 12 occasions by releasing known quantities of marked fish (previously captured in the net) several hundred yards above it and estimating p, the proportion of downstream migrants sampled, by

$$\hat{p}_i = \frac{\text{number recovered}}{\text{number released}}$$

Coho with clipped adipose fins were released above the net on nine occasions and the resulting estimates of p ranged from .03 to .62, with a mean of .33 (Table 7). The release and recovery of sockeye fry on two other dates yielded a very similar mean $\hat{p}$ of .34. A 2 x 11 chi-square test was used to test for significant differences in the proportion of fish released, both sockeye and coho, and subsequently recovered. This test was significant at $\alpha \leq .001$, but the reason for this large variation in estimates could not be ascertained. Since significant day differences in $\hat{p}_i$ existed, the efficiency of the net was estimated by averaging the results from each test:

$$\hat{p} = \frac{\sum_{i=1}^{k} \hat{p}_i}{k}.$$

where k is the number of efficiency experiments.

The variance of $\hat{p}$ was estimated as

$$S_{\hat{p}}^{2} = \sum_{i=1}^{k} \frac{(\hat{p}_i - \bar{\hat{p}}_i)^2}{(k-1)(k)}$$

For each species-age group of interest, the number of outmigrants during a given time period was estimated by dividing the total catch by the efficiency of the net, and multiplying this result by the rate of total days in that time period to the number of days fished (Cochran 1977):

$$\hat{N}_J = \frac{D_J \sum_{i}^{d_J} n_i}{d_J \hat{p}}$$

Table 7. Numbers of coho smolts and sockeye salmon fry released above the net and subsequently recaptured, and the estimated P_i.

Release date	Species	Number released	Number recaptured	$\hat{p}_i$
4/10/81	Sockeye fry	579	276	.48
4/14/81	Sockeye fry	155	33	.21
			Mean	.34
4/30/81	Coho smolts	33	5	.15
5/05/81	Coho smolts	31	1	.03
5/06/81	Coho smolts	28	10	.36
5/12/81	Coho smolts	24	11	.46
5/13/81	Coho smolts	16	4	.25
5/14/81	Coho smolts	13	7	.54
5/15/81	Coho smolts	10	2	.20
5/16/81	Coho smolts	12	2	.17
5/17/81	Coho smolts	10	5	.50
5/18/81	Coho smolts	8	5	.62
			Mean	.33

where

J is a subscript denoting time period,

D_J is the number of days in the time period,

d_J is the number of days fished,

n_i is the catch on day i for a particular species-age grouping,

$\hat{N}_J$ is the estimated total number of outmigrants during that time period.

During the early portion of the outmigration period, when catches were small, a time period length of one month was used. After March 21, this was reduced to two weeks. The total production was estimated by summing the estimated production of each time period.

$$\hat{N} = \sum_{J}^{13} N_J$$

The variance of N_J may be estimated by (D. Chapman, personal communication)

$$S_{\hat{N}_J}^2 = \left(\frac{D_J \sum^{d_J} n_i}{d_J \hat{p}^2}\right)^2 S_{\hat{p}}^2 + \left(\frac{D_J^2}{d_J \hat{p}^2}\right) \frac{\sum^{d_J} n_i}{d_J} .$$

The variance of the estimate of $\hat{N}$ was then obtained by summing the $S_{\hat{N}}^2$ over all time periods.

Length frequency histograms for coho and cutthroat were constructed (computer program FRG 369, Gales and Swanson 1981) and the mean length of each species computed for all time periods. The means obtained were tested with Duncan's multiple range procedure to determine if significant differences existed between them.

Dates for the analysis of the impact of drought-ending rainstorms in the migrations of fish were obtained by review of precipitation data collected by the National Weather Service and Pitt et al. (1981). A storm was considered significant if the total precipitation accumulated was greater than .15 inches.

7.0 RESULTS

7.1 Gravel Composition

7.1.1 Evaluation of Sampling and Analytical Techniques

Possible sources of bias in obtaining bed composition data for Bear and Kelsey Creeks include operator and analytical error. The former is caused by the mechanical disadvantages of the McNeil sampler and by variability in sampling and technique. The manual sampling method necessarily disturbs the substrate as it is worked into the gravel and some fine sediments are undoubtedly lost as a result. Additionally, some of the suspended material inside the cylinder cannot be retrieved by the plunger. Rocks larger than the core diameter cannot be included in the sample; however, this problem was encountered only at KC5. A new location was sampled as near to the original as possible without causing intersample disturbance when the insertion of the sampler was prevented by coarse material. In the vast majority of the samples, however, the core diameter of the sampler was nearly twice the size of the largest particles.

There was little variation in the methods used to collect and wet-sieve the individual gravel samples. The sample volume and the amount of water used to wash the material through the sieves remained fairly constant for each sample during the study. Particular care was taken to perform the various operations in a consistent manner. It is believed that representative samples of the streambed materials were obtained with this technique.

Adams (1980) demonstrated longitudinal and lateral variability in the bed composition of a riffle area. He found that the greatest variation in fine materials occurred in the direction perpendicular to the stream flow.

The gravel samples obtained in the present study were collected at short intervals along a longitudinal axis, which reduced but did not exclude sampling error caused by spatial variability. Site selection within different stations also suffered from this deficiency even though the sites were chosen for uniformity on the basis of flow and surface gravel characteristics.

Some researchers have excluded large rocks from particle distribution analyses, purportedly to reduce the variance in the estimate of percent fines (McNeil and Ahnell 1964, Adams 1980). Large rocks become problematic when a freeze core method is used since they often extend beyond the frozen front of the core (Adams and Beschta 1980). Nevertheless, using the freeze core method to obtain his samples, Adams (1980) compared percent fines values calculated from samples including rocks larger than 50.8 mm with values from samples where large rocks were excluded and concluded that "it seems to make little difference whether one includes or excludes larger rocks when calculating percent fines. The vast majority of the samples had no rocks larger than 50.8 mm and there was no difference in the two criteria." Similarly, none of the samples analyzed in this study contained rocks larger than the 50.8 mm size class, and it was decided to retain all data in order to better quantify the data using statistical techniques.

Perhaps the most serious drawback of the manual core method is its inability to detect stratification of sediments. Vertical heterogeneity has been observed in some spawning materials (Peterson 1978, Shirazi et al. 1979, Adams 1980) but not in others (Platts et al. 1979). For the purposes of this study, the bed composition was assumed to be homogeneous to a depth of 30 cm when in fact this is probably not true (Lotspeich and Reid 1980).

Percent fines was designated as the fraction of bed material less than

0.841 mm in diameter. Several particle sizes have been suggested as a definition of percent fines; however, a standard measure has not been agreed upon. The 0.841 mm size used in this study was selected based on extensive studies over the years at the Fisheries Research Institute. This value has been found to represent those sizes of inorganic sediments that have influence on fish and insect life.

Recently, Platts et al. (1979) and Shirazi et al. (1979) have advocated the use of the mean geometric diameter (d_g) of the particle size distribution as a standard measure of the quality of bed materials in relation to salmonid egg survival and development. These authors noted a strong correlation between d_g and estimates of the permeability and porosity of the stream bed. The question of which variable, d_g or percent fines, is a better descriptor of substrate quality will be deferred until later. It was anticipated that the two variables would be closely correlated in the samples collected in this study. Figure 18 is a plot of percent fines against mean geometric diameter for samples from both streams. It is apparent that the two variables are more strongly associated in Bear Creek ($r^2 = 0.83$) than in Kelsey Creek ($r^2 = 0.44$) gravel samples, which implies that the particle distribution in the latter samples may deviate slightly from the expected log normal distribution. Furthermore, the fitted lines describing the relation between d_g and percent fines intersect in the particle size range of interest, suggesting that in substrates with a small d_g Kelsey Creek has lower levels of fines than Bear, whereas for larger d_g values the pattern is reversed. In order to determine whether this relationship is statistically different for the two streams, the curves were linearized and tested for equal intercepts and slopes. The null hypotheses H_0: $B_0 = B_0$ and H_0: $B_1 = B_1$ were accepted at a 0.05

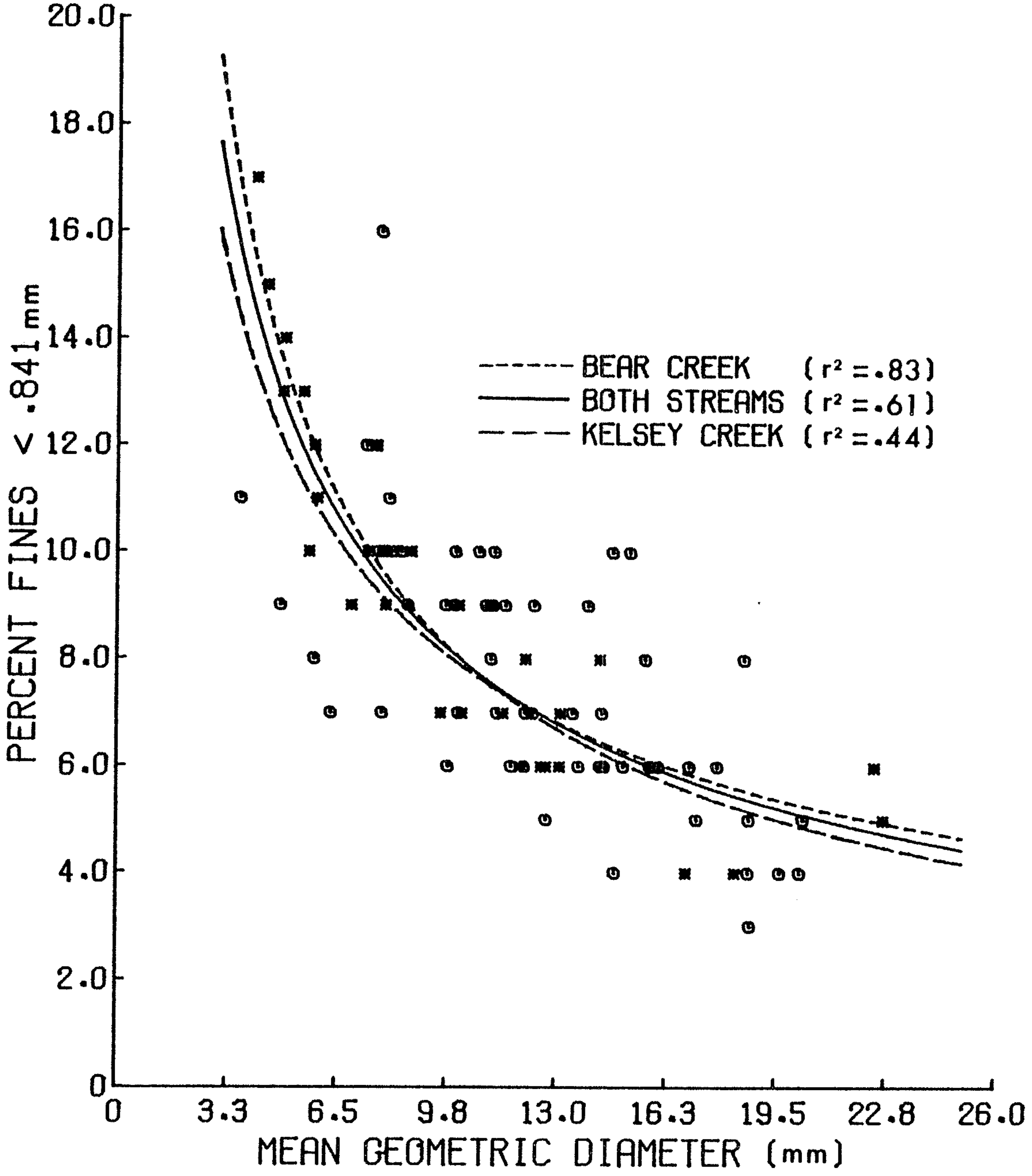


Figure 18. Relationship between percent fines (dry volume, calculated by Shirazi-Seim regression technique) and the mean geometric diameter of the sampled sediment. (ⓣ Kelsey Creek, ✳ Bear Creek).

significance level and we conclude from consideration of both tests that the lines are coincident. Therefore, the curve fitted to the combined data set will suffice to describe both streams and has the equation

$$\log Y = -(0.44 + 0.61 \log X),$$

It should be emphasized that the curves in Figure 18 are of limited predictive value and should not be used to determine a general relationship for substrate materials in other streams.

The percentage of fine sediment in the sample as calculated directly from the sieve data may be expected to differ distinctly from that computed by the regression method of Shirazi and Seim (1979). The percentage of sediment $\leq$ .841 mm in Bear and Kelsey Creek core samples are contrasted in Table 8, where computations are by wet volume, dry volume (wet volume corrected with factors given by Shirazi and Seim 1979), or by a least squares regression. Conversion of wet volume to dry volume "reduced" the percent fines by about 30 percent, while the regression resulted in an additional 25 percent reduction. The reason for the shift in the first case is clear; Shirazi and Seim (1979) found that the volume of water retained by small particles is proportionally greater than that retained by large particles, and this is reflected in the correction factors which they developed.

The least squares methodology has the unfortunate property of systematically underestimating the percentage of fine sediments when the particle size distribution is skewed toward larger particles, as is frequently hypothesized to be true in disturbed watersheds. Under the assumption of a log normal particle size distribution, the regression attempts to linearize the distribution

Table 8. Percent fines in streambed core samples calculated as a percent of the sample wet volume, as a percent of the sample dry volume, and as a percent of dry volume as calculated by the regression technique.

Site	Percent ≤ .841 mm Wet volume	Percent ≤ .841 mm Dry volume	Percent ≤ .841 mm Dry volume - regression
KE1-1	12	8	6
KE1-3	13	9	6
KE1-5	17	12	8
KE2-1	29	21	11
KE2-3	43	32	17
KE2-5	19	13	7
KE3-1	15	11	5
KE3-3	26	19	9
KE3-5	17	12	6
KE4-1	23	17	9
KE4-3	27	20	13
KE4-5	24	19	11
BE1-1	12	7	8
BE1-3	15	11	8
BE1-5	23	16	11
BE2-1	29	22	13
BE2-3	13	9	7
BE2-5	14	10	6
BE3-1	10	6	7
BE3-3	25	17	13
BE3-5	30	22	18
BE4-1	20	14	12
BE4-3	12	8	7
BE4-5	28	22	12
BE4B-1	11	8	6
BE4B-3	17	12	8
BE4B-5	11	7	6
Mean	20	14	9

function of the transformed data. Thus, when a line is fit to a distribution skewed toward larger particles, the percentage of fine particles will be underestimated (Fig. 19). Traditional statistical techniques designed to test the "fit" of an observed distribution are not applicable in this situation as 1) the particle distribution function is in units of weight, not numbers, and 2) the errors introduced at each sieve diameter are not independent.

7.1.2 Substrate Composition Temporal and Spatial Variability, 1979-80

Data necessary to plot the stream profiles shown in Figures 20 and 21 are given in Tables 9, 10, and 11. The means and standard deviations for the percent fines and d_g values are presented for the original data and for the profile analysis data. Attention should be called to the deletion of a single replicate from the BC2 sample collected on March 31, 1980, and the effect it had on the values listed in Table 9. In this case, the replicate d_g values were 22.4, 22.7, and 34.5; the latter value was discarded (Table 10). Consequently, the mean d_g value for Bear Creek samples decreased on this date, concomitant with an increase in the mean percent fines value. Sample variances decreased for both parameters. It is unlikely that the deletion impaired the results of the statistical tests applied to the modified data.

The null hypotheses tested by profile analysis and the appropriate conclusions of the statistical tests are given in Table 11. It can be seen that the results are similar for the profile analysis of percent fines and d_g parameters and that certain influences are justified. The profiles are parallel, implying that simple stream by date interactions are negligible. This conclusion is valid in spite of the cross-over of d_g line segments

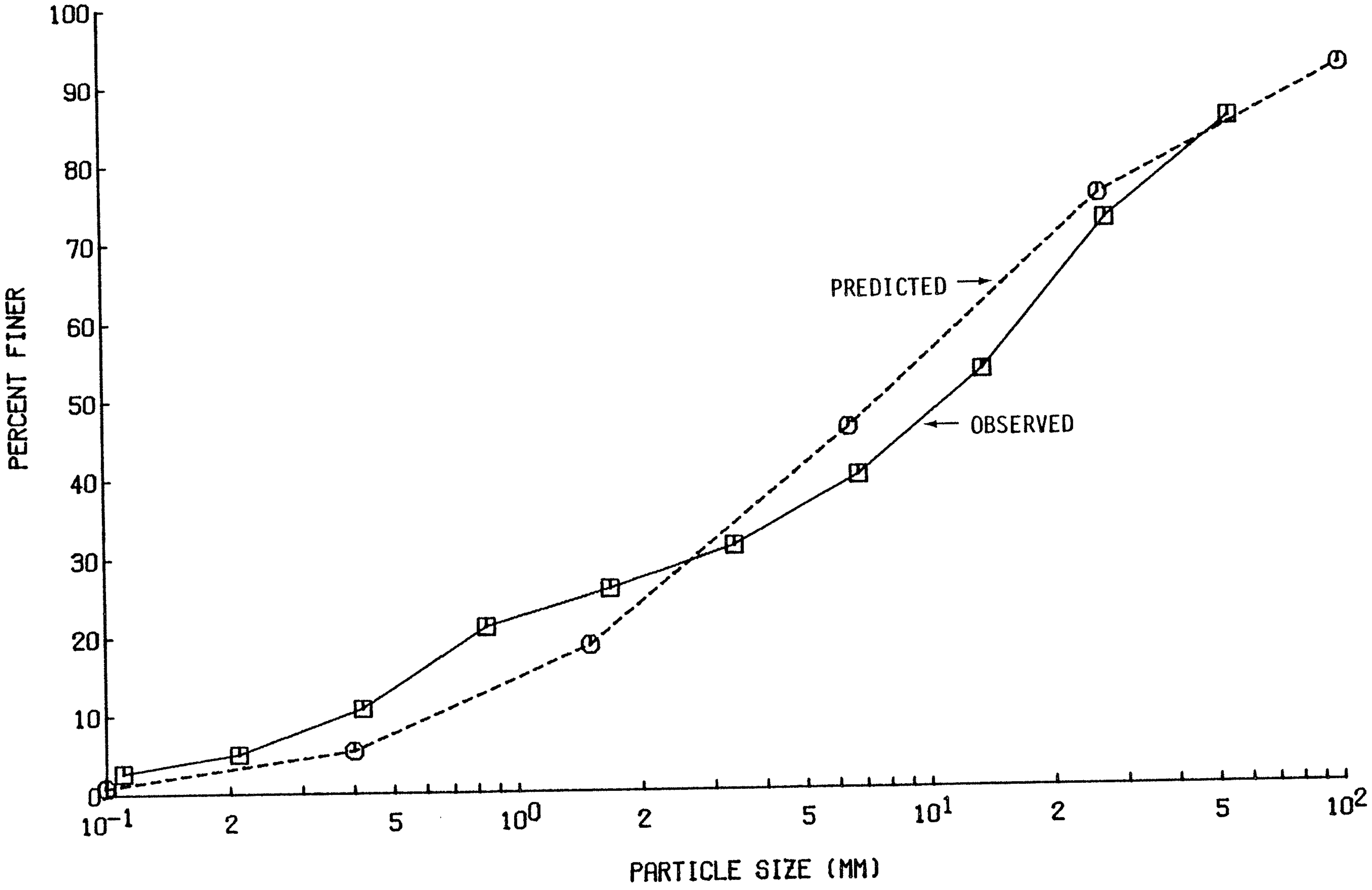


Figure 19. Observed and predicted (by Shirazi-Seim regression technique) particle size distribution of a typical sample from Kelsey Creek.

Table 9. Percent fines (< .841) for replicate samples taken from stations KC1-KC5 and BC1-BC3 in 1979-80.

	Stations							
	KC1	KC2	KC3	KC4	KC5	BC1	BC2	BC3
June 26, 1979	0.10	0.10	0.06	0.05	0.07	0.12	0.08	0.10
	0.09	0.10	0.06	0.07	0.09	0.12	0.08	0.10
	0.08	†	†	0.06	†	†	†	0.09
September 24, 1979	0.09	0.10[a]	0.06	0.10	0.08	0.10	0.09	0.13
	0.12	0.09	0.06[a]	0.09	0.10[a]	0.11	0.12[a]	0.13
	0.06	0.11	0.05	0.06	0.10	0.14	0.07	†
December 28, 1979	0.06	0.04	0.04	0.03	0.07	0.04	0.04	0.06[a]
	0.09[a]	0.07	0.06	0.04	0.07	0.06	0.06[a]	0.07
	0.07	0.06	0.05[a]	0.04[b]	0.04	0.07	0.06	0.07
March 31, 1980	0.09	†	0.05	0.11[a]	0.09	0.17	0.06	0.10
	0.06	0.10	0.06	0.08	0.16	0.09	0.04[a]	0.09[a]
	0.09	0.08	0.07[a]	0.07	0.10	0.15	0.05	0.06

† Samples lost due to collection or analysis errors.

a Samples deleted for profile analysis.

b Sample discarded as outlier.

Table 10. Mean geometric diameter for replicate samples taken from stations KC1-KC5 and BC1-BC3 in 1979-80.

	Stations							
	KC1	KC2	KC3	KC4	KC5	BC1	BC2	BC3
June 26, 1979	10.7	8.4	15.0	20.3	13.5	7.65	14.3	8.7
	12.3	11.2	17.8	14.3	9.7	7.61	12.1	11.1
	15.6	†	†	15.8	†	†	†	11.0
September 24, 1979	13.9	8.1[a]	11.7	7.9	18.5	7.4	10.1	4.9
	7.3	11.1	13.7[a]	8.6	14.6[a]	5.9	5.8[a]	5.5
	16.9	8.0	17.2	14.4	10.0	4.9	13.1	†
December 28, 1979	14.4	20.2	14.7	18.8	12.3	16.9	18.3	12.7[a]
	4.8[a]	11.2	14.3	19.6	10.1	16.9	11.9[a]	9.6
	12.1	12.0	12.7[a]	32.7[b]	18.7	11.4	13.1	10.2
March 31, 1980	10.0	†	18.7	3.6[a]	11.5	4.1	22.4	5.7
	16.0	7.7	9.8	5.8	7.8	7.9	34.5[a]	6.9[a]
	10.9	11.1	7.8[a]	6.3	15.2	4.4	22.7	15.9

† Samples lost due to collection or analysis errors.

a Samples deleted for profile analysis.

b Sample discarded as outlier.

Table 11. Results of profile analysis of substrate samples taken on four dates (refer to Tables 9 and 10) from Kelsey and Bear Creeks, 1979-1980.

	Percent fines		Mean geometric diameter	
Test of parallelism				
Null hypothesis: profiles are parallel				
Observed F with 3 and 15 degrees of freedom	1.92	*accept*	1.60	*accept*
Probability	0.17		0.23	
Test of identical profiles				
Null hypothesis: profiles are identical				
Observed T (two-tailed) with 17 degrees of freedom	-2.1	*accept*	2.1	*accept*
Probability	0.05		0.06	
Test of flat profiles				
Null hypothesis: profiles are flat (i.e., slopes = 0)				
Observed F with 3 and 17 degrees of freedom	25.0	*reject*	3.8	*reject*
Probability	0.00		0.03	
Test of equal mean vectors				
Null hypothesis: mean vectors are equal				
Observed F with 4 and 14 degrees of freedom	2.0	*accept*	2.9	*accept*
Probability	0.17		0.23	

Simultaneous 95.0% Confidence Intervals

		Percent fines			Mean geometric diameter		
Level	Date	Difference between means	Upper limit	Lower limit	Difference between means	Upper limit	Lower limit
1	6/26/79	-0.021	-0.054	0.012	3.37	-2.46	9.19
2	9/24/79	-0.025	-0.069	0.019	4.94	-1.44	11.30
3	12/28/79	-0.004	-0.031	0.022	1.89	-4.55	8.33
4	3/31/79	-0.011	-0.078	0.055	-0.98	-11.90	9.90

between December and March. In contrast, the level of fines in the stream bed is higher in Bear Creek than in Kelsey Creek on all sampling dates (Fig. 20). Parallelism implies that the bed composition on both streams is governed by similar physical factors and that the magnitude of the response is largely the same.

The second hypothesis tested in the analysis of profiles rejected the notion (at $\alpha = 0.10$) that the profiles were on the same level (Table 11). The characteristics of the substrate environment in Kelsey Creek are apparently distinct from those of Bear Creek. From a consideration of the profile data, it may be noted that the greatest differences occur during the summer months. This observation is corroborated by the results of analyses of variance which tested the unmodified data for differences in paired sample means for each date (Table 11). Significant differences were found between the two streams during June and September for both d_g and percent fines. The means for the December samples are close enough to accept the null hypothesis H_0: $\mu_{BC} = \mu_{BC}$. The convergence of the profiles during the winter months is a conspicuous feature of Figures 20 and 21. March samples are characterized by large variances which contributed to the insignificant difference observed for sample means.

The somewhat obvious conclusion that the slopes of the profiles are not equal to zero is provided for by the test results of the third hypothesis (Table 11). Variation over time is particularly evident during the period of high flows, when a large change in the bed composition of both streams occurs.

Analysis of variance and multiple comparison techniques were used to examine the temporal variability indicated by profile analysis, to identify spatial differences in bed composition within each stream, and to gain further

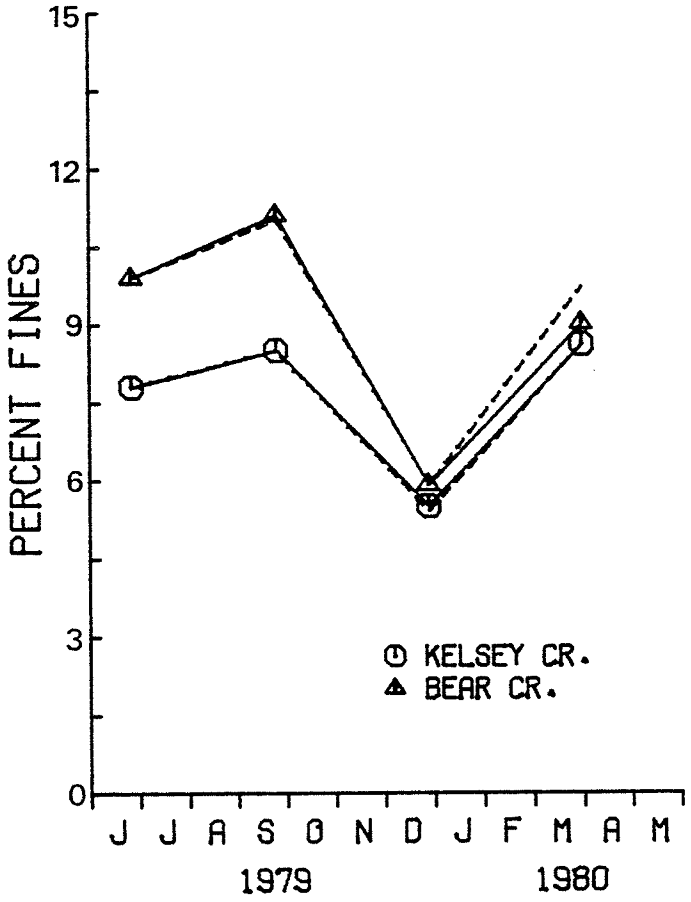


Figure 20. Percentage of fines < .841 mm (dry volume, calculated by Shirazi-Seim regression technique) in Kelsey and Bear Creek substrate samples over time. (Dotted lines indicate all samples included; see text for complete explanation).

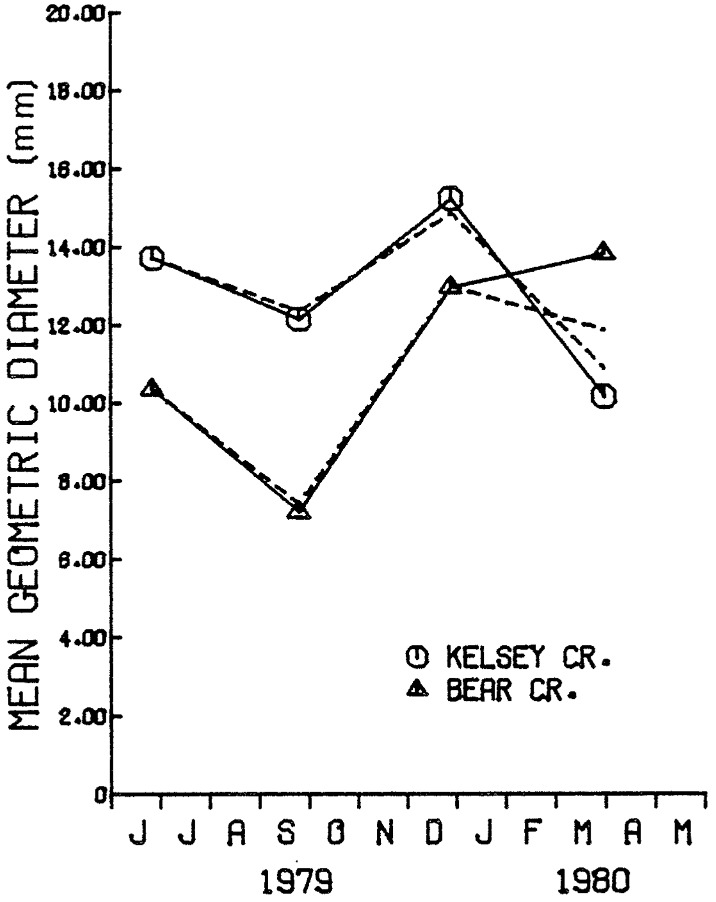


Figure 21. Mean geometric diameter of Kelsey and Bear Creek substrate samples over time. (Dotted lines indicate all samples included; see text for complete explanation).

insight into differences between Kelsey and Bear Creeks. Results of the various ANOVA tests and associated a priori contrasts are summarized in Table 12. The results of the a posteriori comparisons, using the Scheffe procedure, are presented in Figure 22.

Mean percent fines and d_g values differ significantly between stations and dates, as evidenced by the two-way ANOVA results. Clearly, the streambed composition is influenced by these two factors, implying that variability occurs over time and space. The presence of an interaction effect between sampling location and date indicates that the relationship among the bed composition parameters associated with different stations varies according to the date on which the samples are obtained. The joint influence of both factors is primarily in the same direction, i.e., the magnitude of the response observed in the sample means is not the same for all stations. This becomes apparent from an inspection of the mean values presented in Table 9. An example is the disproportionately large decrease in the mean percent fines obtained at KC2 during December. In a few cases the direction of the difference between sample means for a particular station on successive dates is opposite or reversed from the direction of the difference between the corresponding sample means for all other stations. The marked increase in the mean geometric diameter of the bed material collected at BC2 during March may be cited as an example. This contrasts with the observed decrease in d_g values in the rest of the samples obtained on the same date. Any statements about instream spatial variability must therefore by qualified since interaction effects make it difficult to interpret general differences in bed composition between stations.

One-way ANOVA's (Table 12) were used to further distinguish between-

Table 12. Results of the ANOVA tests and associated *a priori* contrasts conducted on Kelsey and Bear Creek substrate samples.

Statistical Procedure[1]	Null Hypothesis	Variable	F ratio	T ratio[2]
1. Two-way ANOVA; Kelsey and Bear Creek stations.	1) H_o: No main effect of station	Fines	9.36***[3]	
	(H_o: $\mu_{KC1} = \mu_{KC2} = \cdots = \mu_{BC3}$)	D_g	6.66***	
	2) H_o: No main effect of date	Fines	20.93***	
	(H_o: $\mu_1 = \mu_2 = \mu_3 = \mu_4$)	D_g	4.97**	
	3) H_o: No station x date interaction	Fines	2.15**	
		D_g	3.28**	
2. One-way ANOVA: Kelsey and Bear Creek stations.	1) H_o: $\mu_{KC1} = \mu_{KC2} = \cdots = \mu_{BC3}$	Fines	4.52***	
		D_g	3.81**	
3. One-way ANOVA: Kelsey Creek stations	1) H_o: $\mu_{KC1} = \mu_{KC2} = \cdots = \mu_{KC5}$	Fines	4.34**	
		D_g	0.75 NS	
3-1 A priori contrasts; Kelsey Creek stations.	1) H_o: $\mu_{KC1} = \mu_{KC5}$	Fines		-0.53 NS
		D_g		-0.39 NS
	2) H_o: $\frac{\mu_{KC1} \quad \mu_{KC2}}{2} = \frac{\mu_{KC4} + \mu_{KC5}}{2}$	Fines		1.03 NS
		D_g		-1.32 NS
	3) H_o: $\frac{\mu_{KC1} + \mu_{KC2} + \mu_{KC5}}{3} = \frac{\mu_{KC3} + \mu_{KC4}}{2}$	Fines		-3.86***
		D_g		1.45 NS
4. One-way ANOVA; Bear Creek stations.	1) H_o: $\mu_{BC1} = \mu_{BC2} = \mu_{BC3}$	Fines	4.99*	
		D_g	9.90***	
4-1 One-way ANOVA; Bear Creek stations.	1) H_o: $\mu_{BC1} = \mu_{BC3}$	Fines		1.29 NS
		D_g		-0.42 NS
	2) H_o: $\frac{\mu_{BC1} + \mu_{BC3}}{2} = \mu_{BC2}$	Fines		2.88**
		D_g		4.43***

[1] Model I ANOVA's with unequal replication.

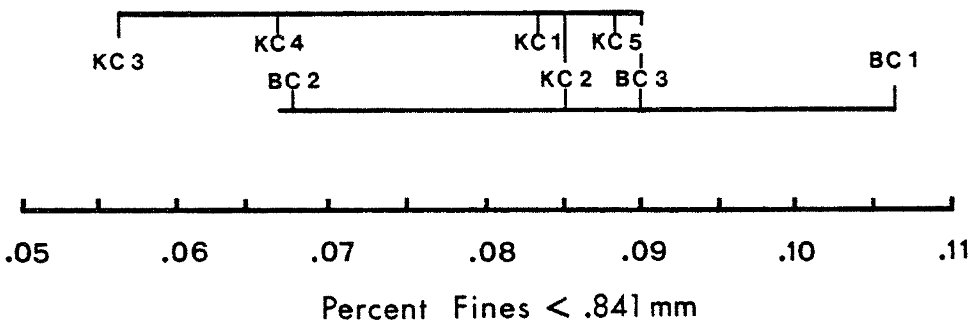
[2] Based on pooled variance estimate.

[3] * = $\alpha \leq .05$

** = $\alpha \leq .01$

*** = $\alpha \leq .001$

NS = nonsignificant



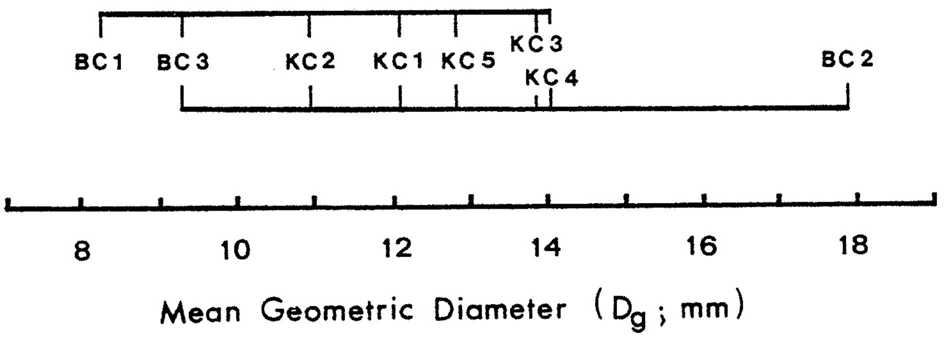


Figure 22. Schematic representation of sample means for Kelsey and Bear Creek stations grouped into homogeneous subsets by the Scheffe method. Ranges are for the .05 level.

station variations in Kelsey and Bear Creek. The null hypotheses of equal parameter means among the different stations of the separate streams were rejected at the 95 percent level, with the exception of mean geometric diameter values in Kelsey Creek. A priori contrasts were appropriate regardless of the outcome of the preliminary ANOVA and the results do not support the general hypothesis that streambed materials become progressively finer in a downstream direction. No significant difference was detected by multiple comparisons of the upper- and lowermost stations in Kelsey and Bear Creeks. However, additional tests reveal that substrates in the middle reaches of both streams are characterized by lower levels of percent fines and a higher mean geometric diameter than the upper and lower sections.

Using the Scheffe method of a posteriori testing, the combined stations of both study streams were grouped into hetergeneous subsets, where the differences in means of any two groups are not significant at the .05 significance level. The subsets are illustrated by the overlapping lines in Figure 22. A pair of means not enclosed by the range of any one line is significantly different. The purpose of making these comparisons was to determine the nature of the variability of substrate characteristics at different locations in the same stream. As in the one-way ANOVA's and their associated a priori tests, the results of Scheffe's method do not entirely account for the influence of sampling date on individual stations. The means portrayed in Figure 22 are the averages for the separate stations computed from all sampling dates and thus represent a general ranking of stations from the entire year's data. On any given sampling date the order may be different from that presented here.

There are no clear boundaries between sets of means not significantly different from each other; the subsets overlap along most of their ranges.

The stations lying in the non-overlapping regions are of particular interest, as are the relative positions along the scale of stations from the same stream on the lines. Looking at the percent fines subsets first, we see that KC3 and BC1 are the lowest and highest ranking stations, respectively. It has already been established that KC3 and KC4 are significantly different from the other Kelsey Creek stations (*a posteriori* tests are generally less sensitive than *a priori* tests) which are grouped closely together in the range of 8 to 9 percent fines. Bear Creek stations appear in the order of BC2 < BC3 < BC1. Although Bear Creek has a fewer number of stations, the range between sample means for Bear Creek stations is slightly greater than the range of Kelsey Creek sample means. The same observation is even more apparent from the d_g ranges shown in Figure 22. The positions of BC1 and BC3 relative to BC2 along the scale reflect the differences noted earlier from *a priori* comparisons. The d_g sample means of all Kelsey Creek stations are grouped together, and were not found to differ significantly under any of the statistical tests employed. KC3 and KC4 obtain the highest average d_g values; however, note that the ordering of Kelsey Creek stations on the d_g scale is not the same as that on the percent fines scale. A simple inverse relationship between percent fines and mean geometric diameter is not apparent from Kelsey Creek samples.

7.1.3 Variables Affecting Bed Composition

Although it has been noted that changes in bed composition are associated with temporal and spatial variability, it is important to point out that sampling date and station are not proximal factors which determine substrate composition. More precisely, they represent a complex and interrelated set of environmental effects acting on specific areas of the stream bed. Any attempt to identify sources of variability in bed composition over space and time must provide for more immediate causal factors, such as hydraulic, biological, and geomorphic phenomena. The stepwise regression methodology employed in this study sought to quantify the relationship between the substrate parameters, percent fines and mean geometric diameter, and the physical and hydraulic variables listed in Table 13. These variables represent only a subset of the possible factors which influence bed composition and, therefore, the derived regression models are limited in their comprehensiveness and predictive ability.

Correlation coefficients between the selected variables and average percent fines and d_g values are shown in Table 13. It can be seen that the two dependent variables are strongly correlated with each other, which is a relationship already discussed. Higher correlations generally exist between percent fines and the independent variables than between d_g and the same variables. It is perhaps surprising that the downstream distance (DIST) and channel sinuosity (SINU) are negatively correlated with percent fines since the amount of fines in the stream bed was expected to increase as these measures increased. The correlation coefficients between independent variables, particularly those relating to stream discharge, indicate that the variables are not all independent of one another at the sampling sites studied.

The results of the stepwise regression analyses are given in Table 14. The first set of results for each bed composition parameter (Models 1a and 2a) are the significant regression equations obtained using a significance level of 0.05 as the criteria for entering and removing a variable from the model (F .05 (1,29) = 4.18).

The second set of results is the stepwise/equations resulting from the

Table 13. Correlation matrix for dependent and independent variables (c.f. Table 4) for Kelsey and Bear Creeks.

Variables	D_g	Q	MAXQ	MINQ	INSTQ	PRECIP	GRAD	DIST	SINU
FINES	-0.73	-0.35	-0.40	-0.06	-0.20	0.36	-0.55	-0.03	-0.28
D_g		0.18	0.18	0.16	0.13	-0.24	0.44	0.08	0.18
Q			0.95	0.78	0.94	-0.48	-0.20	0.22	-0.24
MAXQ				0.62	0.80	-0.47	-0.09	0.30	-0.14
MINQ					0.86	-0.44	-0.26	0.30	-0.31
INSTQ						-0.44	-0.27	0.18	-0.31
PRECIP							0.00	0.00	0.01
GRAD								0.14	0.58
DIST									0.45

Table 14. Results of the stepwise regression analyses for the Kelsey and Bear Creek substrate samples.

Step (1)	Equation (2)	Explained Variance R^2 (3)	F-value (4)
(Model 1a)	Percent Fines; critical F = 4.18 (x = 0.05)		
1	Fines = 0.11 - 1.01 GRAD	0.307	13.3
2	Fines = 0.12 - 1.18 GRAD - 5.60 x 10^{-4} Q	0.529	16.3
(Model 1b)	Percent Fines; critical F = 2.10 (x = 0.10)		
1	Fines = 0.11 - 1.01 GRAD	0.307	13.3
2	Fines = 0.12 - 1.18 GRAD - 5.60 x 10^{-4} Q	0.529	16.3
3	Fines = 0.12 - 1.10 GRAD - 8.99 x 10^{-4} Q + 1.63 x 10^{-3} MINQ	0.583	13.1
4	Fines = 0.11 - 1.06 GRAD - 8.15 x 10^{-4} Q + 1.80 x 10^{-3} MINQ + 9.02 x 10^{-4} PRECIP	0.615	10.8
5	Fines = 0.11 - 1.14 GRAD - 1.75 x 10^{-3} Q + 2.68 x 10^{-3} MINQ + 1.00 x 10^{-3} PRECIP + 2.60 x 10^{-4} MAXQ	0.643	9.4
6	Fines = 0.10 - 1.13 GRAD - 3.85 x 10^{-3} Q + 2.17 x 10^{-3} MINQ + 9.45 x 10^{-4} PRECIP + 5.69 x 10^{-4} MAXQ + 2.96 x 10^{-3} INSTQ	0.711	10.2
(Model 2a)	Mean Geometric Diameter; critical F = 4.18 (α = 0.05)		
1	D_g = 7.91 + 145.94 GRAD	0.191	7.1
(Model 2b)	Mean Geometric Diameter; critical F = 2.10 (α = 0.10)		
1	D_g = 7.91 + 145.94 GRAD	0.191	7.1
2	D_g = 5.60 + 171.47 GRAD + 0.23 MINQ	0.271	5.4

selection of variables under the condition that $\alpha = 0.10$ ($F_{0.10\ (1,29)} = 2.10$). The consequence of using less stringent criteria was to allow more variables to enter and remain in the models; it did not alter the sequence of variable selection. In each case, the graphical and statistical analysis of residuals did not indicate any violation of assumptions necessary for the development of a regression model. The final regression model associated with the specified conditions includes those variables present at the last step of analysis.

As indicated in Table 14, the hydraulic gradient measured at each sampling site is the most important variable affecting percent fines and mean geometric diameter of the bed composition. The gradient alone accounts for 31 and 19 percent of the observed variation in percent fines and d_g, respectively, and is the sole variable to enter the d_g regression model when the significance level is 0.05. Model 2b incorporates one additional variable, MINQ, which represents the minimum discharge recorded during the month previous to sampling. Note that only 27 percent of the variation in d_g is explained by these two variables, an improvement of 8 percent over Model 2a.

In conjunction with hydraulic gradient, the average discharge, Q, for the month preceding sample collection is significantly related to the amount of fines observed in the stream bed. The final regression equation accounts for 53 percent of the variation in the dependent variable. Model 1b includes all of the variables related to discharge as well as GRAD and PRECIP (the number of rain-free days prior to sampling). These variables are retained in spite of evidence that colinearities are present. Approximately 71 percent of the variance in percent fines measurements is explained by the presence of six independent variables at the 0.10 significance level.

7.1.4 Gravel Composition, Spring 1981

The percentage of sediment (dry volume) < .841 mm, as determined from streambed core-samples taken at each site in early April, 1981, is shown in Table 15. Site averages for Kelsey Creek ranged from 10 percent at KE1 to 22 percent at KE2. The average (by site) percent fines in Bear Creek had a range of approximately one-half that observed in Kelsey Creek. The maximum average volume of fine sediment, 15 percent, occurred at BE3, while the minimum average of 9 percent was found at BE4B. Statistical tests showed no difference in the percent fines among sites within a stream or between streams. A U-test comparing the percentage of fine sediment in March 31, 1980, samples from Bear and Kelsey Creeks with those obtained on April 13, 1981, (percent fines calculated by regression method in each case) was not significant. Apparently no gross changes in streambed composition occurred during 1980.

7.2 Streambed Stability

On several instances incorrect scour chain excavation procedures resulted in estimates of negative scour; that is, the chains were found at a greater elevation than that at which they were left after the previous sampling date. Data collected from scour chain sites at which this occurred were not included in the analysis which follows.

The depth of streambed scour in Kelsey Creek varied both temporally and spatially, with a range extending from 0-18 cm (Table 16). Scour was linearly related to discharge at the mouth of Kelsey Creek at 4 out of the 7 sites monitored (KE3, KE4 (Fig. 23); KS2, KS3 (Fig. 24). Site KE1 exhibited a convex curvilinear relationship (Fig. 23). The remaining two sites, KE2 (Fig. 23) and KS1 (Fig. 24), were not exposed to appreciable scour. Slope coefficients

Table 15. The percentage (dry volume) of fine sediment (≤ .841 mm) in core samples obtained during the week of April 13, 1981, and the concentration of oxygen in interstitial water sampled at that site on March 19 or 20, 1981. (n.s. = not sampled).

		% fine sediment (D.O. mg/l)		
		Standpipe Number		
Stream	Site	1	3	5
Kelsey Creek				
	KE1	8.1 (n.s.)	8.8 (n.s.)	12.2 (3.0)
	KE2	21.1 (3.2)	32.2 (1.9)	13.4 (6.8)
	KE3	10.8 (6.7)	18.9 (4.3)	11.7 (1.8)
	KE4	16.9 (6.5)	20.1 (3.5)	18.8 (3.3)
Bear Creek				
	BE1	7.0 (n.s.)	10.9 (n.s.)	16.4 (8.8)
	BE2	21.7 (7.8)	8.9 (9.5)	10.0 (8.8)
	BE3	6.5 (n.s.)	16.9 (n.s.)	22.0 (5.7)
	BE4	14.0 (n.s.)	8.1 (7.8)	21.5 (6.0)
	BE4B	7.6 (n.s.)	12.4 (n.s.)	7.3 (n.s.)

Table 16. Mean scour at transect sites on Kelsey Creek and peak discharge during each time period as measured at the U.S.G.S. gaging site on Kelsey Creek.

Site	Time period	Peak discharge (m^3/sec)	Mean scour (cm)
KE1	12/06/80-12/29/80	11.0	—
	12/29/80-02/16/81	3.9	15.5
	2/16/81-03/20/81	2.8	18.0
	3/20/81-04/25/81	2.1	1.2
	4/25/81-07/09/81	2.8	—
	7/09/81-08/11/81	8.1	16.8
KE2	12/06/80-01/10/81	11.0	1.5
	1/10/81-01/31/81	3.9	4.9
	1/31/81-03/24/81	2.8	6.1
	3/24/81-04/25/81	1.9	3.0
	4/25/81-07/09/81	2.8	—
	7/09/81-08/03/81	8.1	3.4
KE3	12/06/80-12/29/80	11.0	11.9
	12/29/80-01/31/81	3.9	0.9
	1/31/81-03/24/81	2.8	7.6
	3/24/81-04/25/81	1.9	1.8
	4/25/81-07/09/81	2.8	3.0
	7/09/81-08/03/81	8.1	5.8
KE4	12/06/80-01/10/81	11.0	7.6
	1/10/81-01/31/81	3.9	—
	1/31/81-03/24/81	2.8	4.6
	3/24/81-04/25/81	1.9	3.7
	4/25/81-07/09/81	2.8	2.7
	7/09/81-08/03/81	8.1	9.1
KS1	12/22/80-01/17/81	11.0	0.6
	1/17/81-02/21/81	3.9	5.8
	2/21/81-03/19/81	2.8	3.0
	3/19/81-04/25/81	2.1	1.5
	4/25/81-07/09/81	2.8	1.2
	7/09/81-08/03/81	8.1	2.4
KS2	12/22/80-01/17/81	11.0	—
	1/17/81-02/21/81	3.9	7.6
	2/21/81-03/19/81	2.8	3.7
	3/19/81-04/25/81	2.1	1.5
	4/25/81-07/09/81	2.8	—
	7/09/81-08/03/81	8.1	17.7
KS3	12/22/80-01/17/81	11.0	9.8
	1/17/81-02/21/81	3.9	4.0
	2/21/81-03/19/81	2.8	2.1
	3/19/81-04/25/81	2.1	0
	4/25/81-07/09/81	2.8	7.9
	7/09/81-08/03/81	8.1	8.2

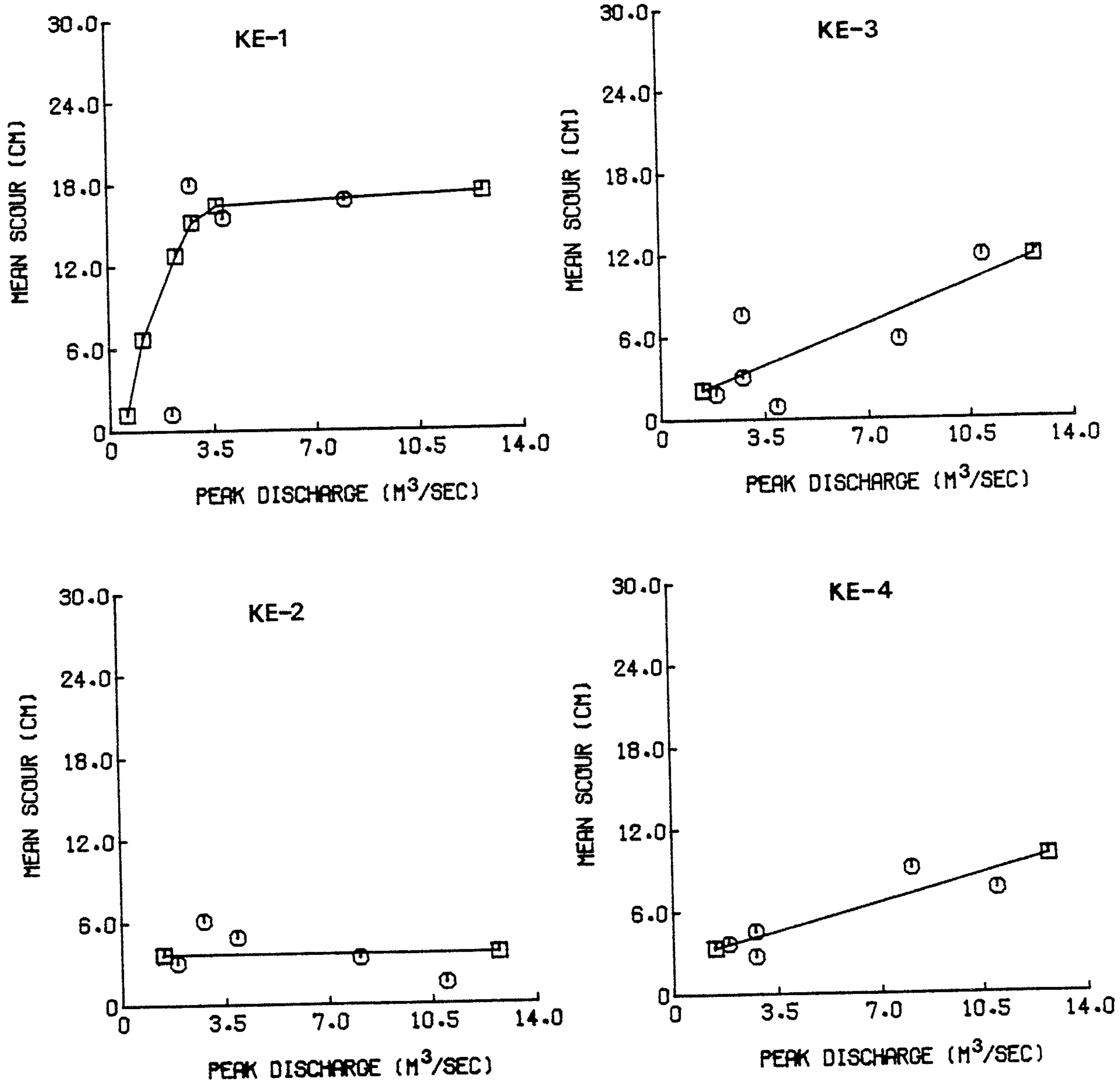


Figure 23. Relationship between streambed scour and discharge at the Kelsey Creek instream embryo bioassay sites KE1 to KE4.

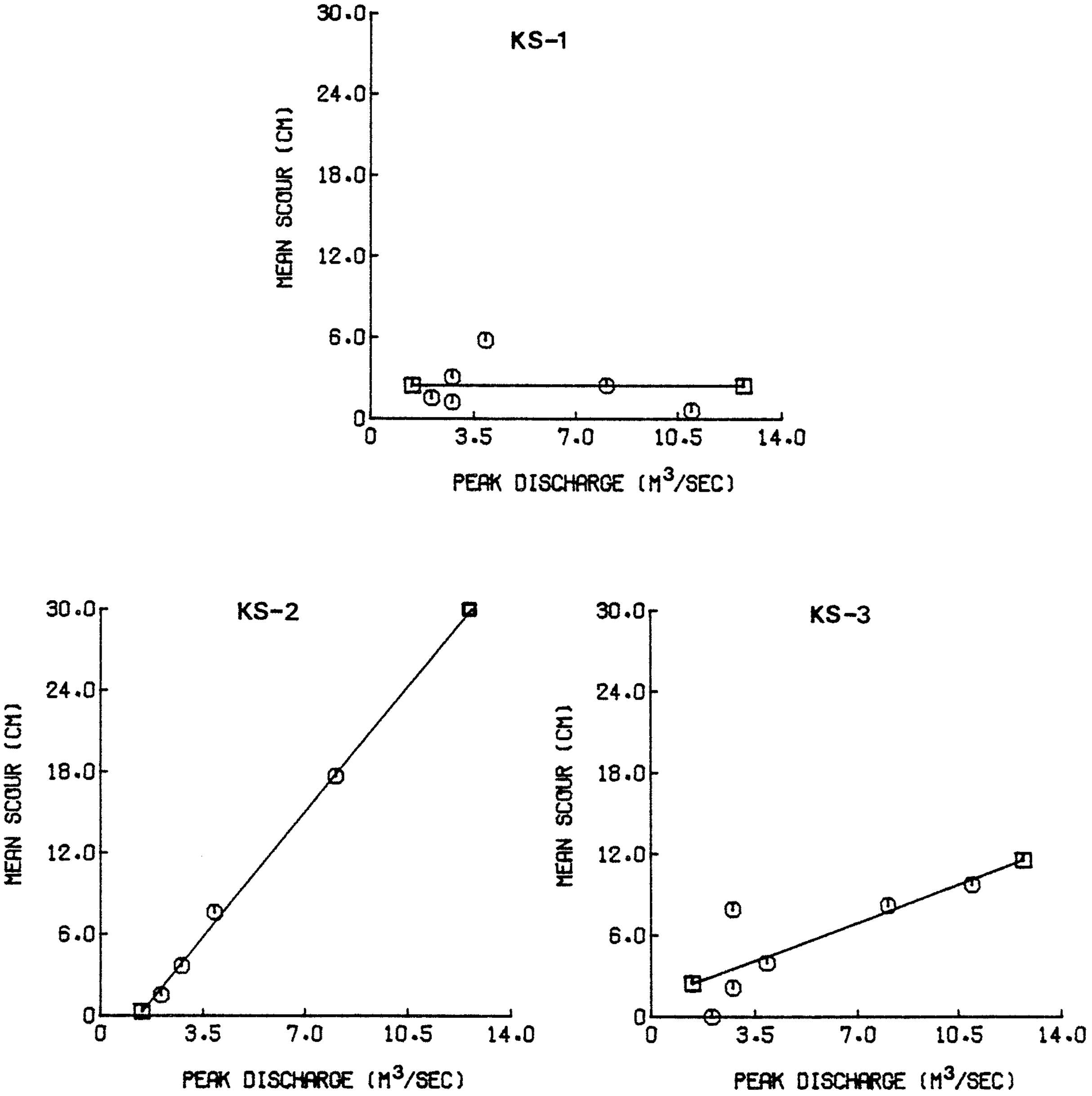


Figure 24. Relationship between streambed scour and discharge at the Kelsey Creek scour sites, KS1 to KS3.

of each of the regression equations differed significantly.

Discharge in Bear Creek during the same time period was insufficient to initiate large-scale bedload movement. The greatest mean site scour was 9 cm (Table 17), though values were typically far less than this. No site on Bear Creek exhibited a significant depth of scour-discharge relationship.

Mean site scour in Bear Creek was significantly less than in the urban stream during two out of the three time periods for which data were collected. In the period December 6, 1980 to March 24, 1981, the peak instantaneous discharge in Kelsey Creek was 10.90 m^3/sec. A Mann-Whitney U-test comparing the mean site scour in each stream during this time was significant at $\alpha \leq .002$. Conversely, during the second time interval (spring bioassays, March 24 to July 9), major rainstorms were absent, and the mean site scour depth did not differ between streams. In the final time period (July 9 to August 3) a summer storm raised the instantaneous peak discharge in Kelsey Creek to 8.07 m^3/sec. Once more, mean site scour in Kelsey Creek was significantly greater than that in Bear Creek ($\alpha \leq .002$).

The estimated percentage of Kelsey Creek streambed scoured to a depth of D_i or greater is shown in Figure 25 for selected discharges (see also Table 18). As could be deduced from the discharge-scour depth regressions presented earlier, an increase in the peak instantaneous discharge results in a greater proportion of the streambed being scoured to a greater depth. Concomitantly, egg mortality due to scour rises (Fig. 26) reaching at a discharge of 12.73 m^3/sec, 18 and 48 percent for coho and cutthroat, respectively.

The impact which the channel disequilibrium in Kelsey Creek has had upon the incubating embryos may be roughly estimated using the relationship between stream discharge and embryo mortality. Spawner surveys were employed to

Table 17. Mean scour at transect sites on Bear Creek and peak discharge during each time period as measured at the U.S.G.S. gaging site on Bear Creek.

Site	Time period	Peak discharge (m^3/sec)	Mean scour (cm)
BE1	12/22/81-12/29/81	3.97	0
	12/29/80-02/01/81	2.87	5.8
	2/01/81-03/24/81	2.58	2.1
	3/24/81-04/25/81	1.36	4.9
	4/25/81-07/09/91	1.05	2.7
	7/09/91-08/11/81	1.57	1.8
BE2	12/22/80-12/30/80	3.97	0
	12/30/80-02/01/81	2.87	2.1
	2/01/81-03/24/81	2.58	2.1
	3/24/81-04/25/81	1.36	0
	4/25/81-07/09/81	1.05	4.6
	7/09/81-08/11/81	1.57	0.6
BE3	12/22/80-12/30/80	3.97	0
	12/30/80-02/16/81	2.87	2.4
	2/16/81-03/24/81	2.58	0.6
	3/24/81-04/25/81	1.36	2.7
	4/25/81-07/09/81	1.05	—
	7/09/81-08/11/81	1.57	2.4
BE4	12/22/80-12/30/80	3.97	2.4
	12/30/80-02/16/81	2.87	0
	2/16/81-03/24/81	2.58	0.6
BS1	12/27/80-01/17/81	3.97	0
	1/17/81-02/21/81	2.58	2.7
	2/21/81-03/24/81	1.36	9.1
	4/25/81-07/09/81	1.05	2.4
	7/09/81-08/11/81	1.57	1.5
BS2	12/27/80-01/17/81	3.97	0.6
	1/17/81-02/21/81	2.58	1.2
	2/21/81-03/24/81	1.67	0.9
	3/24/81-04/25/81	1.36	2.7
	4/25/81-07/09/81	1.05	1.2
	7/09/81-08/11/81	1.57	1.8

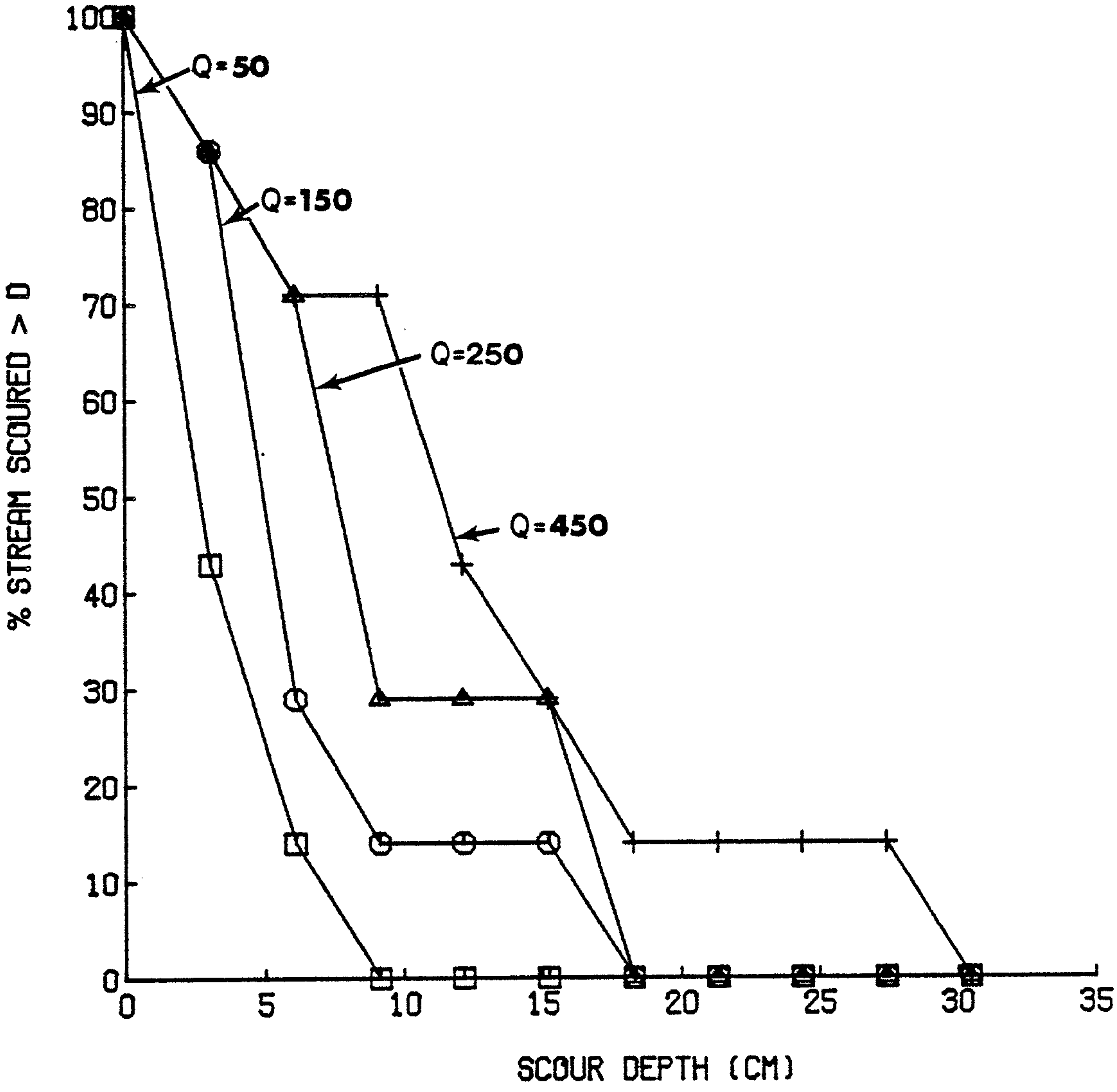


Figure 25. Exceedance curves for the depth of scour in Kelsey Creek at selected discharges.

Table 18. Proportion of Kelsey Creek streambed with scour in the depth interval D_i $[F(D_i)-F(D_{i-1})]$ for discharge increments (Q) of 1,415 m^3/sec, and the probability that a coho salmon or cutthroat trout egg will be buried at a depth of D_i or less $[E(D_i)]$.

	Depth D_i	Proportion of streambed with scour in D_i for each discharge below									$E(D_i)$	
i	(cm)	Q=1.42	Q=2.83	Q=4.24	Q=5.66	Q=7.08	Q=8.49	Q=9.90	Q=11.32	Q=12.74	Coho	Cutthroat
1	0-2.7	.571	.143	.143	.143	.143	.143	.143	.143	.143	0	0
2	2.8-5.8	.286	.714	.571	.429	.143	.143	.143	.143	.143	.011	.015
3	5.9-8.8	.143	0	.143	.143	.429	.429	.143	.143	0	.034	.098
4	8.9-11.9		0	0	.143	0	0	.286	.286	.286	.092	.352
5	12.0-14.9		0	0	0	0	0	0	0	.143	.198	.692
6	15.0-18.0		.143	.143	.143	.286	.143	.143	.143	.143	.358	.921
7	18.1-21.0						.143	0	0	0	.544	.989
8	21.1-24.1							0	0	0	.727	1.000
9	24.2-27.1							.143	.143	0	.860	1.000
10	27.2-30.2									.143	.942	1.000

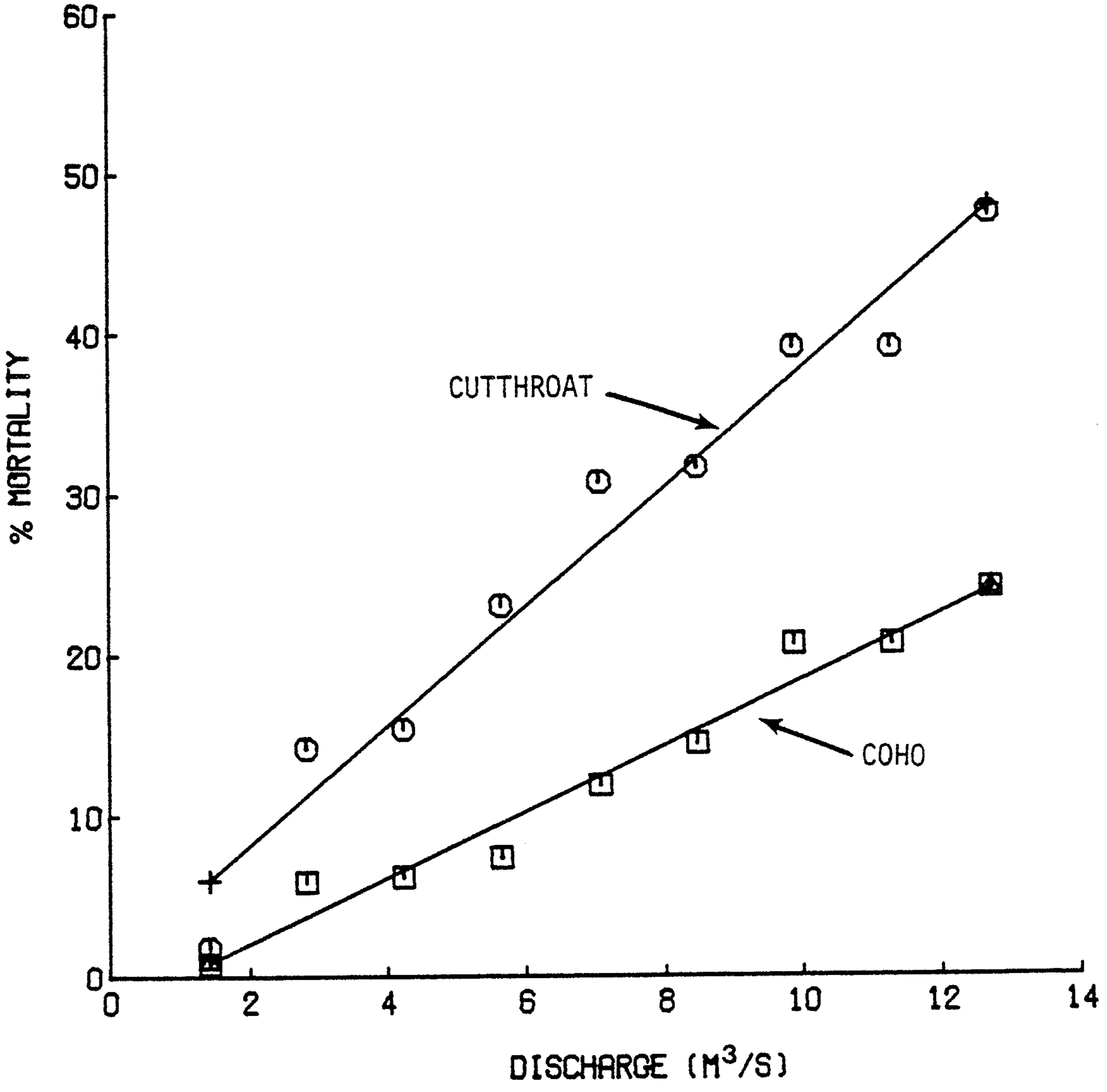


Figure 26. Estimated coho salmon and cutthroat trout embryo mortalities as a function of stream discharge.

estimate the time at which eggs were implanted in the gravel, and the mean date of emergence was roughly estimated from electrofishing data. The mortality of eggs due to scour was then estimated by multiplying the mortality caused by a discharge of given magnitude by the percentage of the embryos which experienced it. The mortality of eggs spawned subsequent to that time was determined only by the peak discharge to which they were exposed. In 1980, coho spawned in Kelsey Creek from October 16 to December 16 (Fig. 27), and the mean date for fry emergence was roughly April 15. Cutthroat spawned from January 15 to April 13 (Fig. 28) and the mean date of fry emergence was about June 10. Note that in applying this method to estimate scour mortality in previous years, it has been assumed that the time of adult spawning, date of mean fry emergence, and the discharge-scour relationship are consistent from year to year.

The mean of the estimated coho salmon embryo scour mortalities in the water years 1978-1981 is 13 percent, with a peak of 23 percent in 1980 (Table 19). The shallower depth of cutthroat redds increases the susceptibility of their embryos to scour, even though peak instantaneous discharges in the spring are generally not as great as during the winter coho embryo incubation period. In the water years 1978-1981, the estimated cutthroat embryo mortality due to scour averaged 14 percent and reached a maximum of 22 percent in 1980 (Table 20).

7.3 Intragravel Water Quality

The intragravel environment in Bear and Kelsey Creeks was distinguished by fluctuating concentrations of dissolved oxygen, pH, and ammonia in 1979-80.

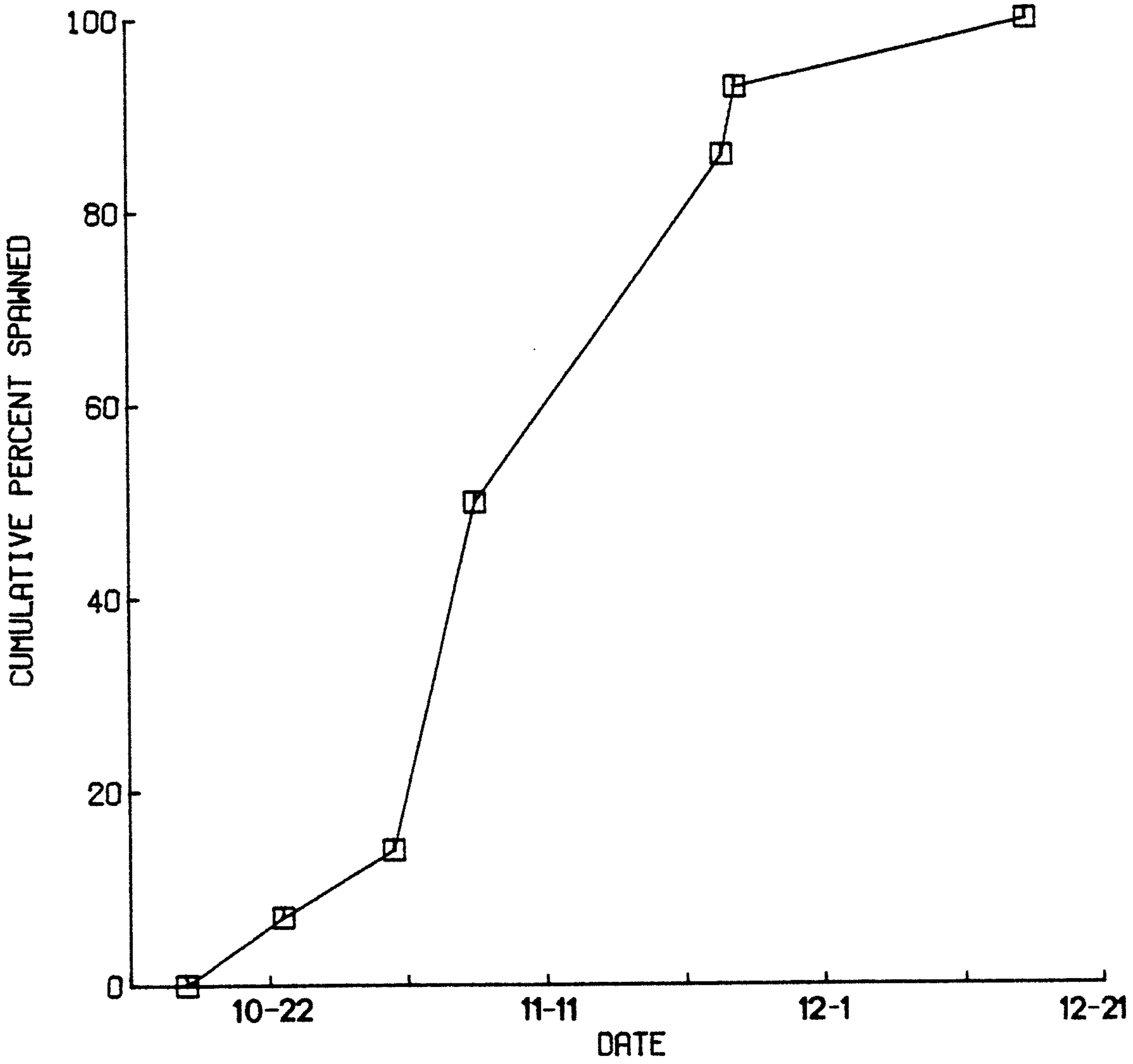


Figure 27. Cumulative percentage of coho salmon spawning in Kelsey Creek, 1980.

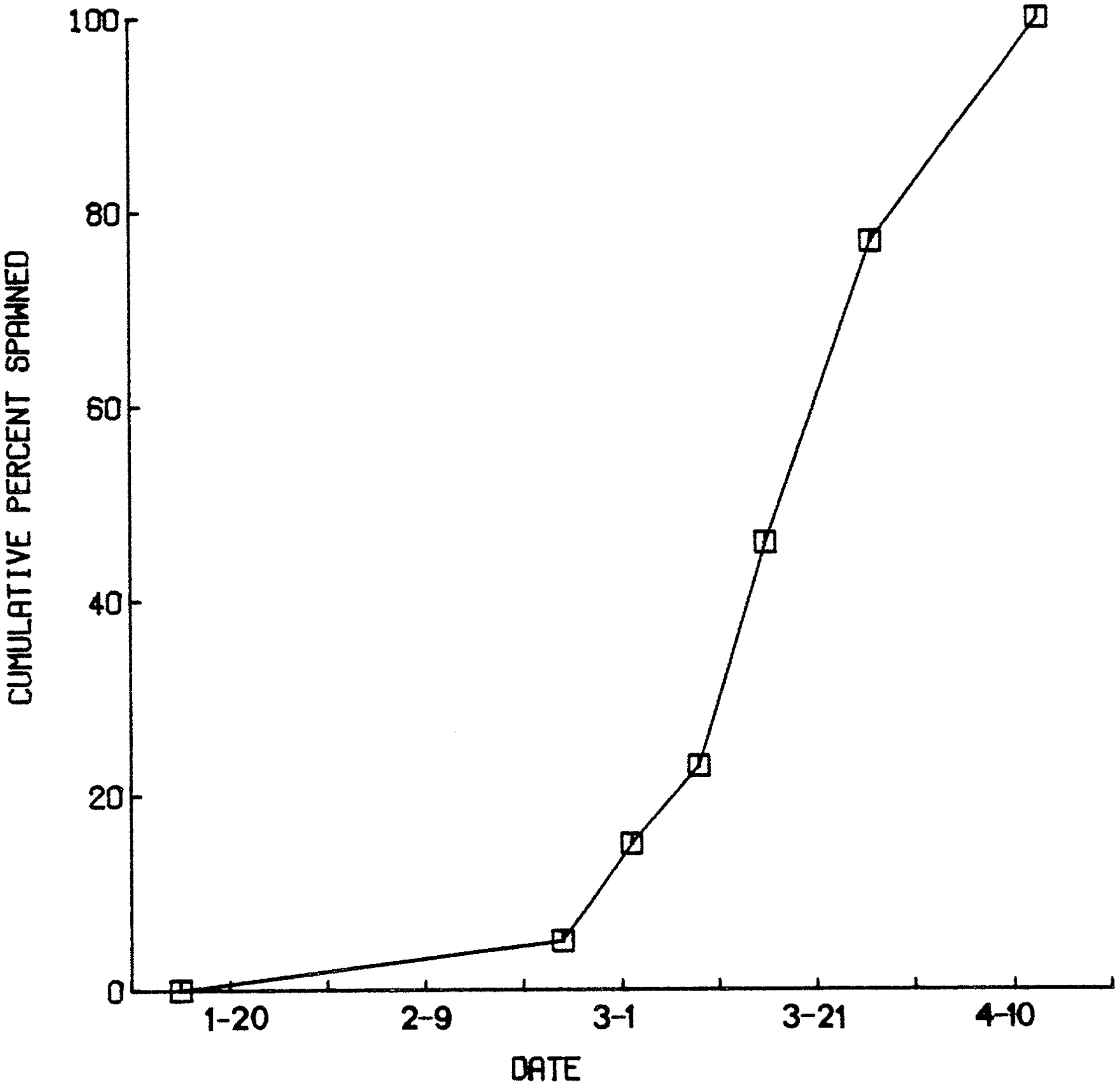


Figure 28. Cumulative percentage of cutthroat trout spawning in Kelsey Creek, 1981.

Table 19. Estimated mortality of coho salmon embryos due to scour, water years 1978-1981.

Water year	Date	% eggs spawned	Q^1 (m^3/s)	Estimated mortality	Total mortality
1978	12/11/77	97	7.6	.07	.07
	12/15/77	100	6.0	<.01	
1979	11/04/78	26	9.8	.03	.05
	2/12/79	100	5.2	.02	
1980	1/12/80	100	14.7	.23	.23
1981	11/21/80	78	11.7	.13	.16
	12/26/80	100	11.0	.03	
				Mean	.127

[1] From U.S.G.S. gauging records.

Table 20. Estimated mortality of cutthroat trout embryos due to scour, water years 1978 to 1981.

Water year	Date	% eggs spawned	Q^1 (m^3/s)	Estimated mortality	Total mortality
1978	4/18/78	100	3.3	.13	.13
1979	4/12/79	100	2.7	.11	.11
1980	4/19/80	100	5.8	.22	.22
1981	5/11/81	100	2.8	.11	.11
				Mean	.142

---

[1] From U.S.G.S. gauging records.

Figure 29 is a plot of the mean values obtained for water quality parameters, excluding un-ionized ammonia for each sampling date. The data used to tabulate the means are given in Appendix 1-3, and the results of statistical tests referred to in this section may be found in Table 21. From the graph it may be inferred that seasonal trends in intragravel dissolved oxygen, pH, and NH_3-N concentrations are similar in both streams. The content of dissolved oxygen in the intragravel water varied significantly over time, with minimum values recorded in August and maximum levels observed in December and January. Mean dissolved oxygen concentrations were comparable in both streams, although Bear Creek measurements were slightly higher during the winter months. Hydrogen ion concentrations ranged from 6.6 to 7.6 in Kelsey Creek and 6.5 to 7.6 in Bear Creek, with minimum and maximum levels occurring in May and September, respectively (Appendix Table 2). Variation of pH by date was found to be significant. Kelsey Creek interstitial samples contained significantly higher concentrations of total and un-ionized ammonia than did Bear Creek samples, often three or four times as much; however, the hypothesis that either form of ammonia varied over time was rejected. Nevertheless, the late summer and autumn months are generally characterized by higher NH_3-N levels than those measured in the remainder of the year.

The measurement of intragravel water temperatures was discontinued after the August 2, 1979, sampling date. Intragravel water temperatures increased slightly in a downstream direction and did not appear to differ from surface water temperatures at sunrise. Maximum and minimum surface water temperatures for Kelsey and Bear Creeks are presented in Figures 30 and 31, respectively.

One-way analysis of variance did not reveal significant differences in

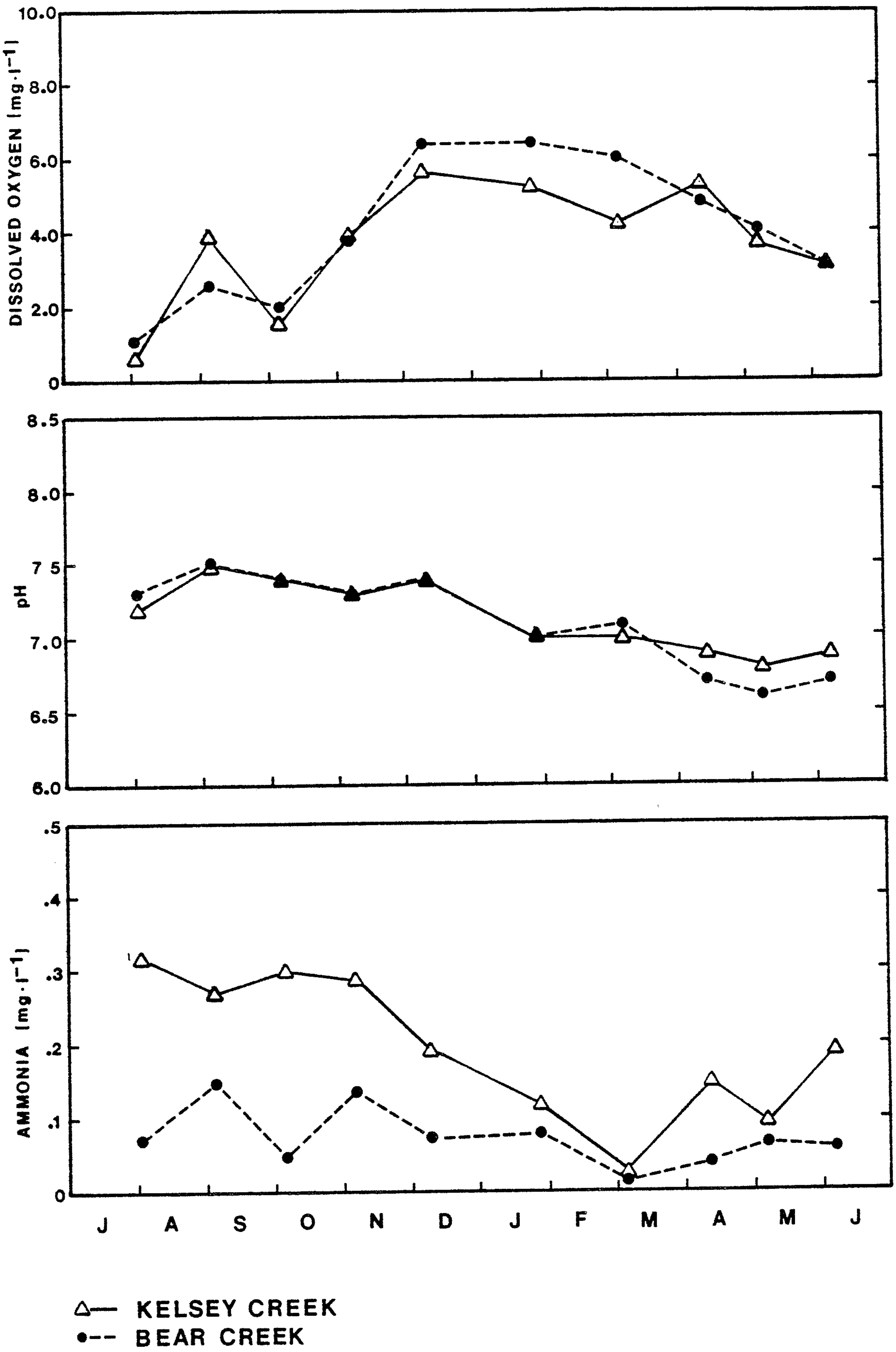


Figure 29. Mean dissolved oxygen concentrations, pH, and ammonia concentrations of interstitial water samples from Bear and Kelsey Creeks, July 1979 to June 1980.

Table 21. The effects of station and sampling date on interstitial water quality samples from Kelsey and Bear Creeks, 1979-80.

Stations	Sample size	Null hypothesis	Variable	F ratios[1] Station	F ratios[1] Date
Kelsey Creek	50	H_0: $\mu_{KC1} = \mu_{KC2} = \dots = \mu_{KC5}$	D.O.	1.79 NS[2]	9.22 ***
			pH	0.32 NS	13.60 ***
			NH_3-N	1.02 NS	1.55 NS
Bear Creek	30	H_0: $\mu_{BC1} = \mu_{BC2} = \mu_{BC3}$	D.O.	0.26 NS	8.04 ***
			pH	0.09 NS	28.86 ***
			NH_3-N	0.51 NS	2.24 NS
Both streams	80	H_0: $\mu_{KC1} = \mu_{KC2} = \dots = \mu_{BC3}$	D.O.	1.07 NS	15.93 ***
			pH	0.23 NS	29.25 ***
			NH_3-N	2.64 **	1.69 NS

[1] Model I ANOVA's with equal replications.

[2] * = $\alpha \leq .05$
** = $\alpha \leq .01$
*** = $\alpha \leq .001$
NS = nonsignificant

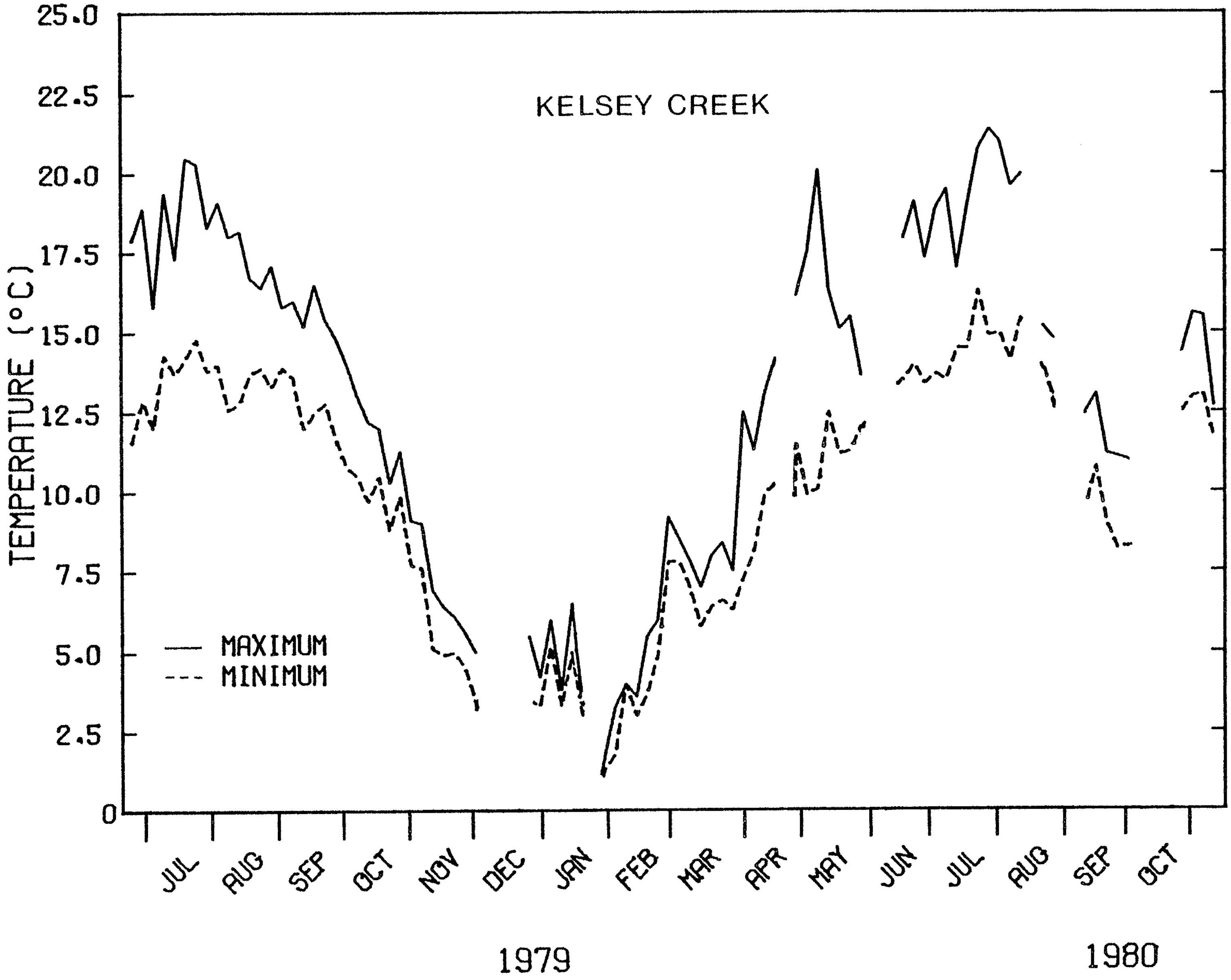


Figure 30. Maximum and minimum temperature of Kelsey Creek, June 1979 to November 1980.

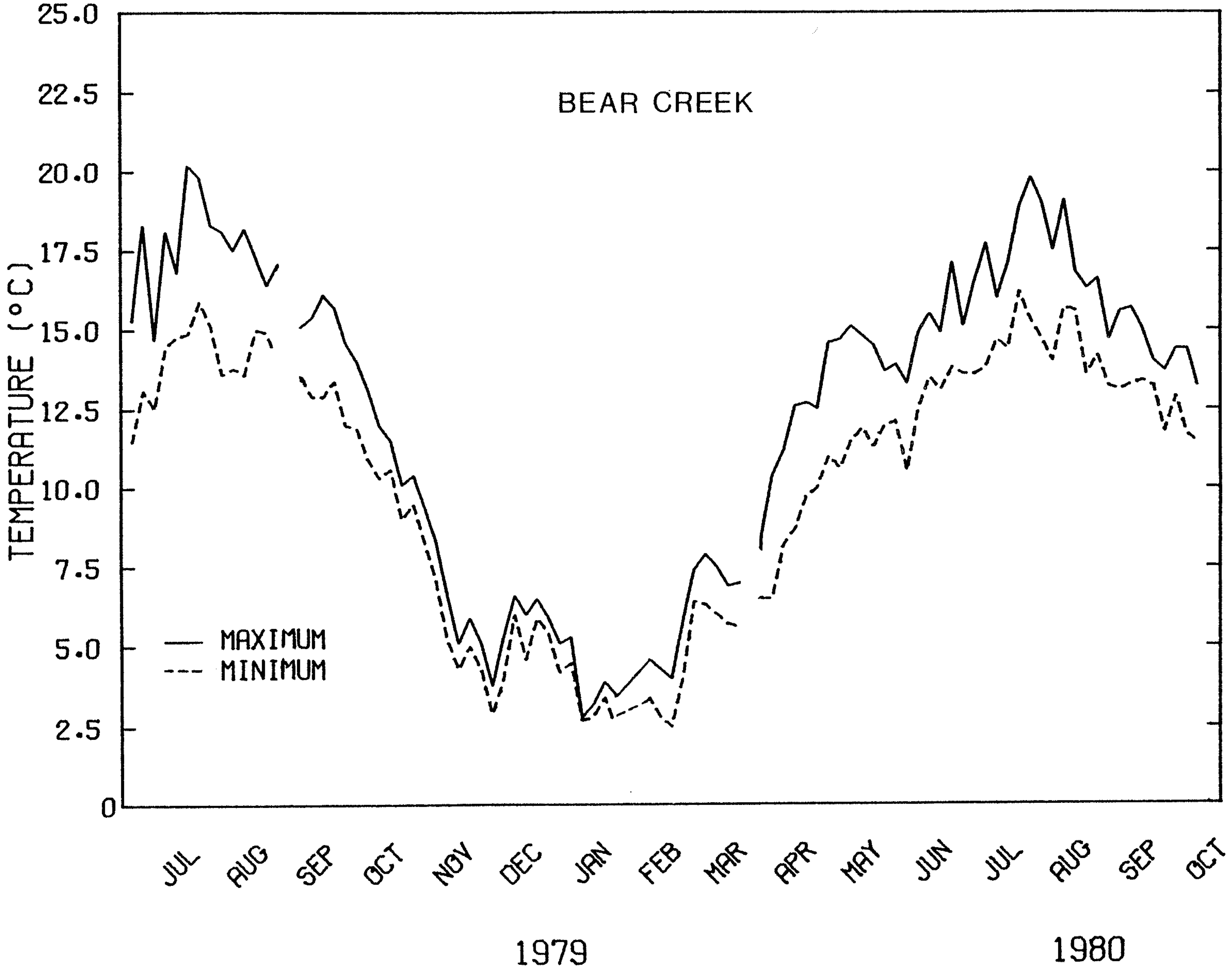


Figure 31. Maximum and minimum daily temperature in Bear Creek, June 1979 to October 1980.

intragravel dissolved oxygen (Appendix Table 3), pH, and total ammonia concentrations between stations in either stream. Consequently, a posteriori analyses of the water quality parameters using Scheffe's method ($\alpha = 0.05$) resulted in a single homogeneous subset, the range of which encompassed the parameter means of all stations. Un-ionized ammonia levels were not included in the analysis since pH and temperature conditions were similar for both streams, and therefore NH_3-N varied proportionately with total ammonia concentrations. The general order of station dissolved oxygen, pH, and total ammonia means, averaged for all sampling dates, is presented here.

Dissolved oxygen ($mg \cdot l^{-1}$)	KC5	KC1	BC2	KC2	BC1	KC3	BC3	KC4
	2.6	3.3	3.7	4.0	4.1	4.2	4.3	4.4
pH	BC1	KC5	BC3	KC4	BC2	KC1	KC3	KC2
	7.1	7.1	7.1	7.1	7.1	7.2	7.2	7.2
Total ammonia ($mg \cdot l^{-1}$)	BC1	BC3	BC2	KC4	KC3	KC1	KC5	KC2
	.06	.07	.09	.13	.16	.18	.23	.28

The order of stations in the dissolved oxygen subset should be compared with the subsets in Figure 22 which were constructed from percent fines and d_g data. Kelsey Creek stations 3 and 4 exhibit the highest dissolved oxygen and d_g means, and the smallest amount of fines. The ranking of Kelsey Creek stations is similar in all of these subsets. In contrast, while BC2 has the lowest percent fines and the highest mean geometric diameter of the Bear Creek stations, it also has the lowest average dissolved oxygen concentration. A possible explanation for this incongruity is that two of the three standpipes at this station were positioned in substrates unlike the stream bed sampled for composition analysis. Water samples from these standpipes may have depressed the station mean dissolved oxygen concentration.

Linear regression analysis was used to further explain the relation between dissolved oxygen levels in the intragravel water at gravel sampling areas and the bed composition parameters. A significant relationship was found between the variables; however, a greater fraction of the total variation in dissolved oxygen concentration was explained by the percentage of fines than by the mean geometric diameter of the gravel samples (Figs. 32 and 33). The separate regression lines derived from Kelsey and Bear Creek data were not significantly different from each other, and may be described by the regression equations

$$\text{D.O. } (\text{mg}\cdot\text{l}^{-1}) = 1.74 + 0.19\, D_g \qquad r^2 = .12$$

$$\text{D.O. } (\text{mg}\cdot\text{l}^{-1}) = 9.08 - 62.76 \text{ fines} \qquad r^2 = .41$$

It may be concluded that percent fines is a better predictor of intragravel water dissolved oxygen concentrations than is mean geometric diameter.

The average intragravel dissolved oxygen concentration at each site on Kelsey Creek (Fig. 34) varied considerably during the winter bioassay period. The range observed extended from 3.53 mg/l at site KE4 on December 5, 1981, to 8.53 mg/l at site KE2 on January 30, 1981. With two exceptions, a statistical test revealed that differences in intragravel dissolved oxygen concentrations within sites were as great as those which occurred between sites. Site differences were apparent only on January 30, 1981, when KE1 and KE2 differed by 3.16 mg/l ($\alpha \leq .043$) and on February 20, 1981, when KE4 and and KE3 differed by 3.00 mg/l ($\alpha \leq .012$).

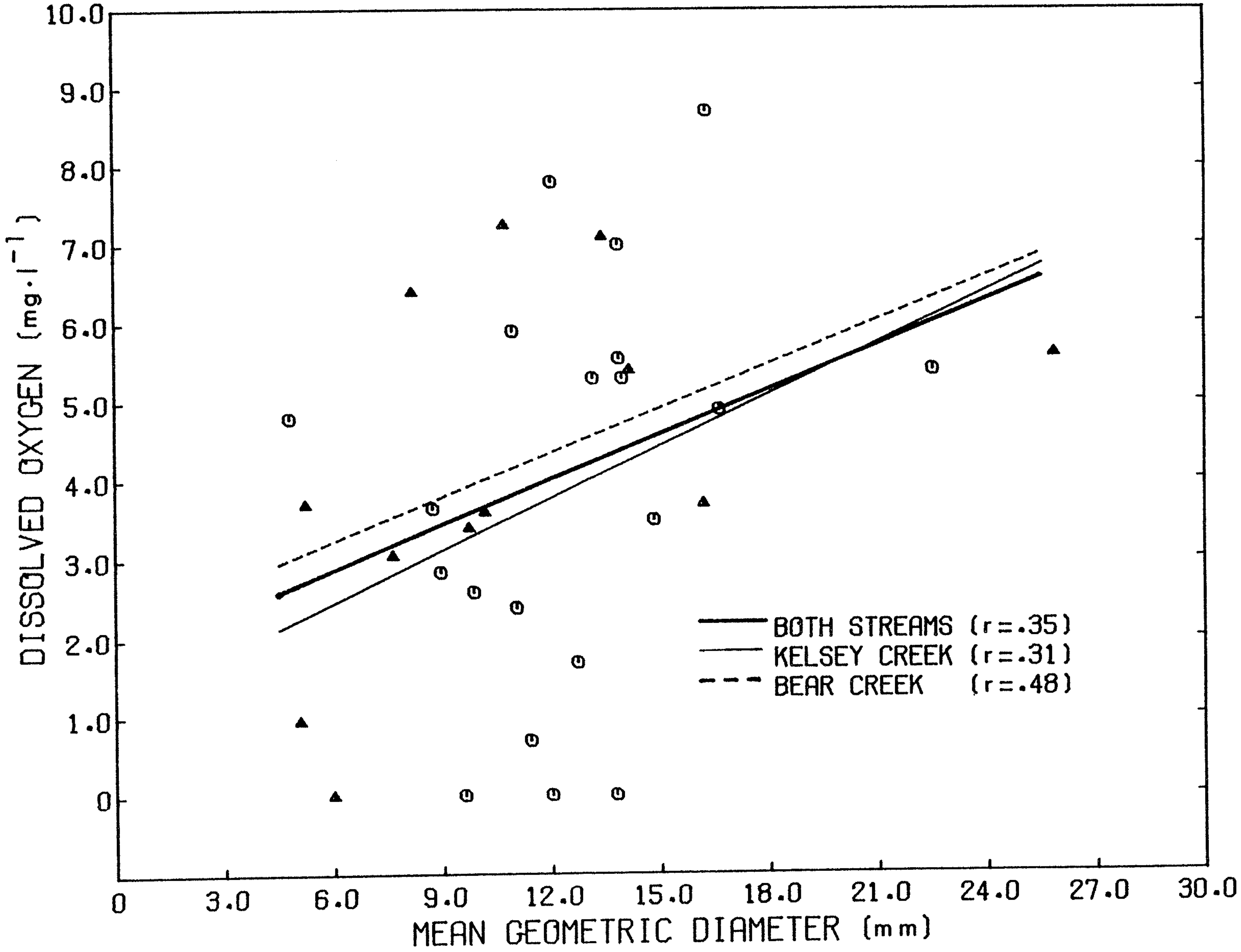


Figure 32. The relationship between the intragravel oxygen concentration and the mean geometric diameter of the substrate sample at each of the Bear (▲) and Kelsey Creek (⊙) sample sites.

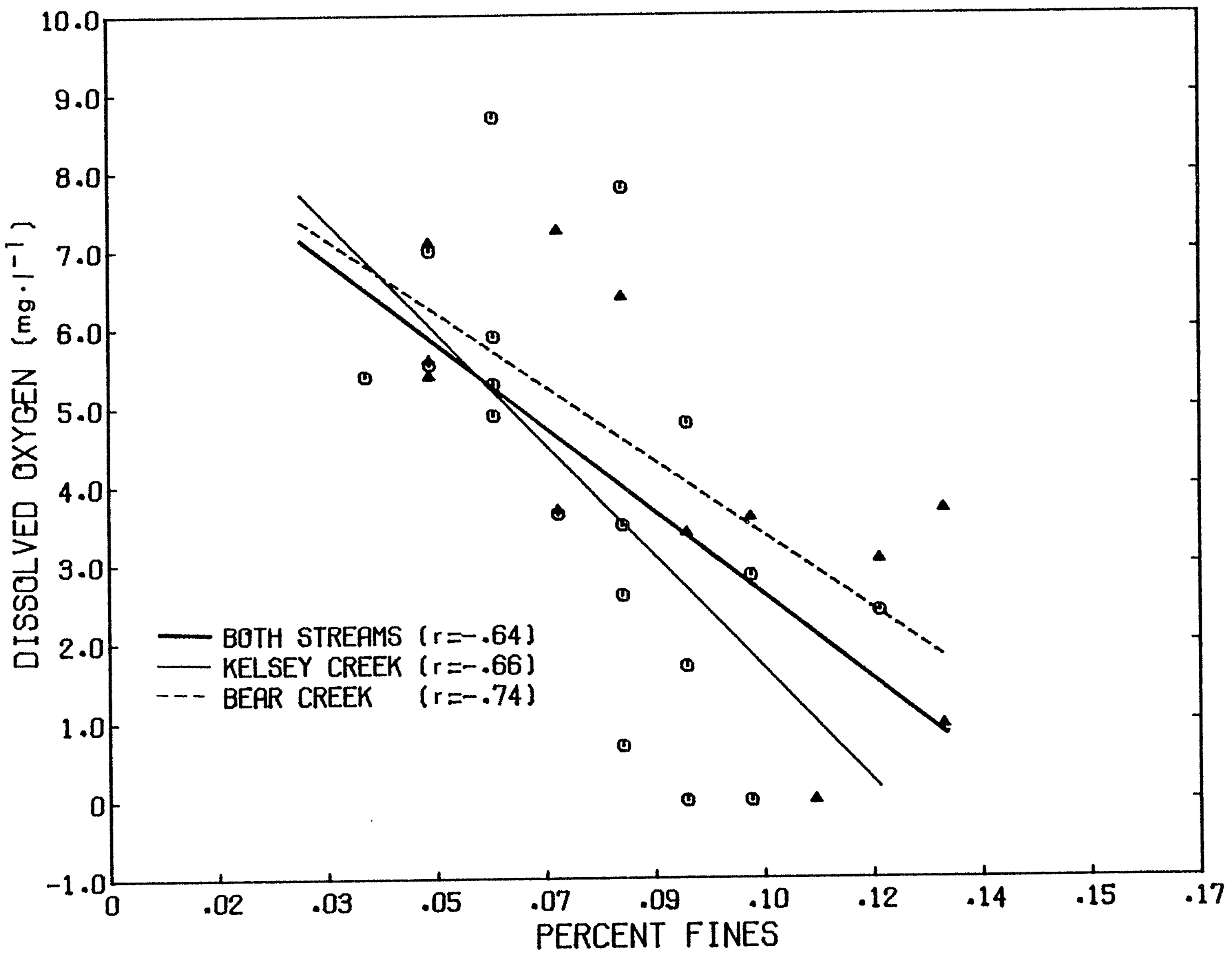


Figure 33. The relationship between the intragravel oxygen concentration and the percentage of fine sediment (dry volume calculated by the Shirazi-Seim regression technique) in the substrate sample at each of the Bear (▲) and Kelsey Creek (ⓞ) sample

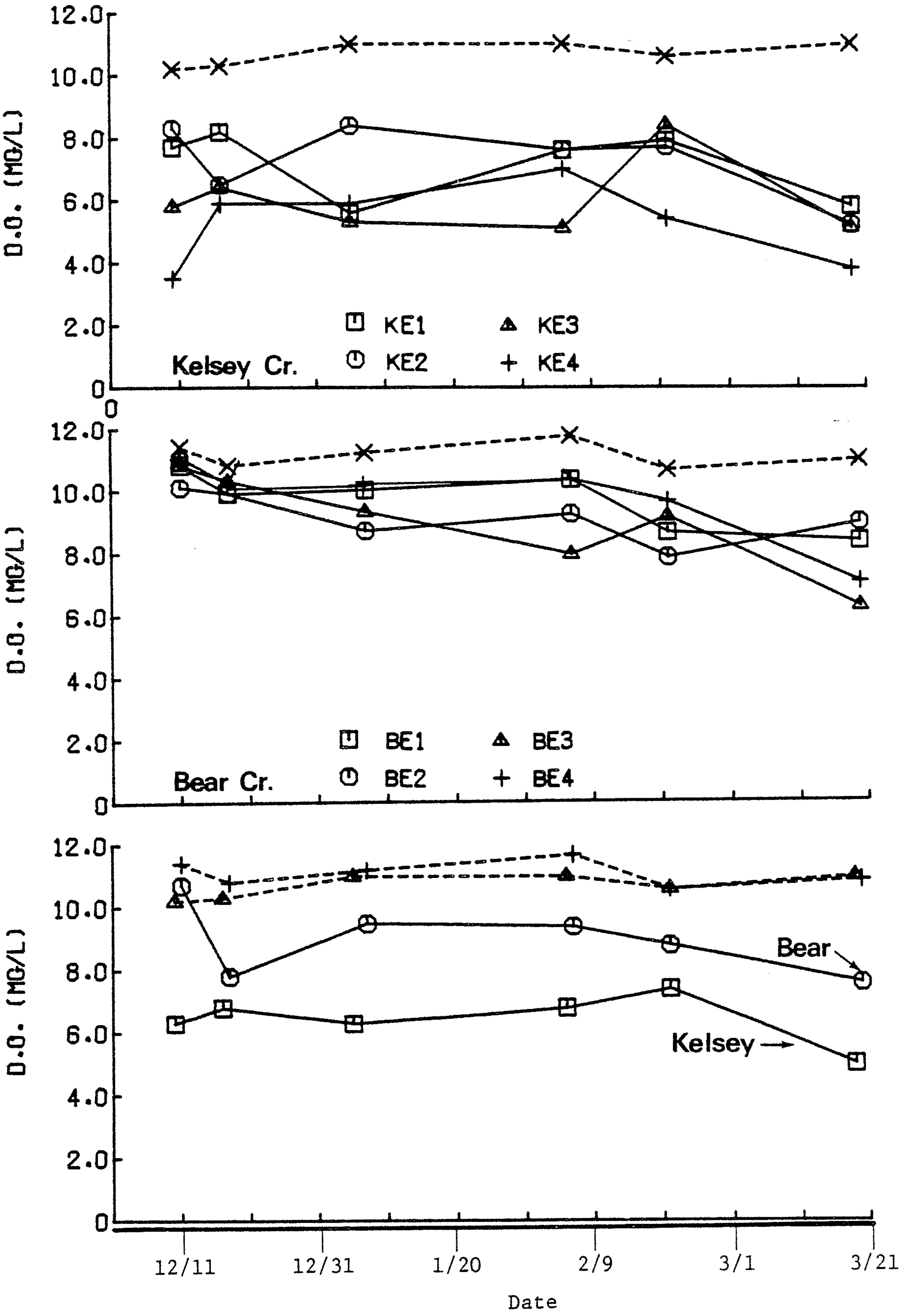


Figure 34. Mean intragravel (solid lines) and surface (dotted lines) water dissolved oxygen concentrations at sites in Kelsey Creek (upper), Bear Creek (middle), and streamwide averages (lower) during the winter instream embryo bioassays.

Intragravel dissolved oxygen concentrations in Bear Creek (Fig. 34) during the same time period were only slightly less variable. Average values per site ranged from 11.1 mg/l at BE4 on December 11, 1981, to 6.2 mg/l at BE3 on March 19, 1981. Sites BE4 and BE3 were significantly different ($\alpha \leq .043$) on December 28, 1980, as were BE4 and BE3 ($\alpha \leq .043$) on February 6, 1981.

These results indicate that intragravel dissolved oxygen levels on each stream on each date can safely be assumed to be homogeneous, and thus a test for significant differences between the pooled data for each stream is appropriate. The intragravel dissolved oxygen concentrations during the winter bioassay period, pooled by stream, are shown in Fig. 34. A Mann-Whitney U-test revealed that concentrations in Bear Creek were significantly greater than those in Kelsey Creek ($\alpha \leq .001$).

Surface water dissolved oxygen concentrations varied little between sites in either stream, and were fairly consistent temporally as well. Concentrations in Kelsey Creek ranged from 10.2 mg/l on December 10, 1981 to 11.0 mg/l on January 3, February 5, and March 18, 1981. In Bear Creek, concentrations were slightly higher ranging from 10.6 mg/l on February 21 to 11.7 mg/l on February 6. The surface waters in both streams were typically 85 to 95 percent saturated with dissolved oxygen.

Intragravel dissolved oxygen concentrations in both Kelsey and Bear Creeks during the spring bioassays (Fig. 35) were lower than those observed during the previous winter. Site averages dropped to as low as .65 mg/l (KE3, May 12) on Kelsey Creek and to 3.1 mg/l on Bear Creek (BE4B, May 26). In general, intragravel dissolved oxygen concentrations showed similar temporal trends at all sites on each stream, and statistically, differences

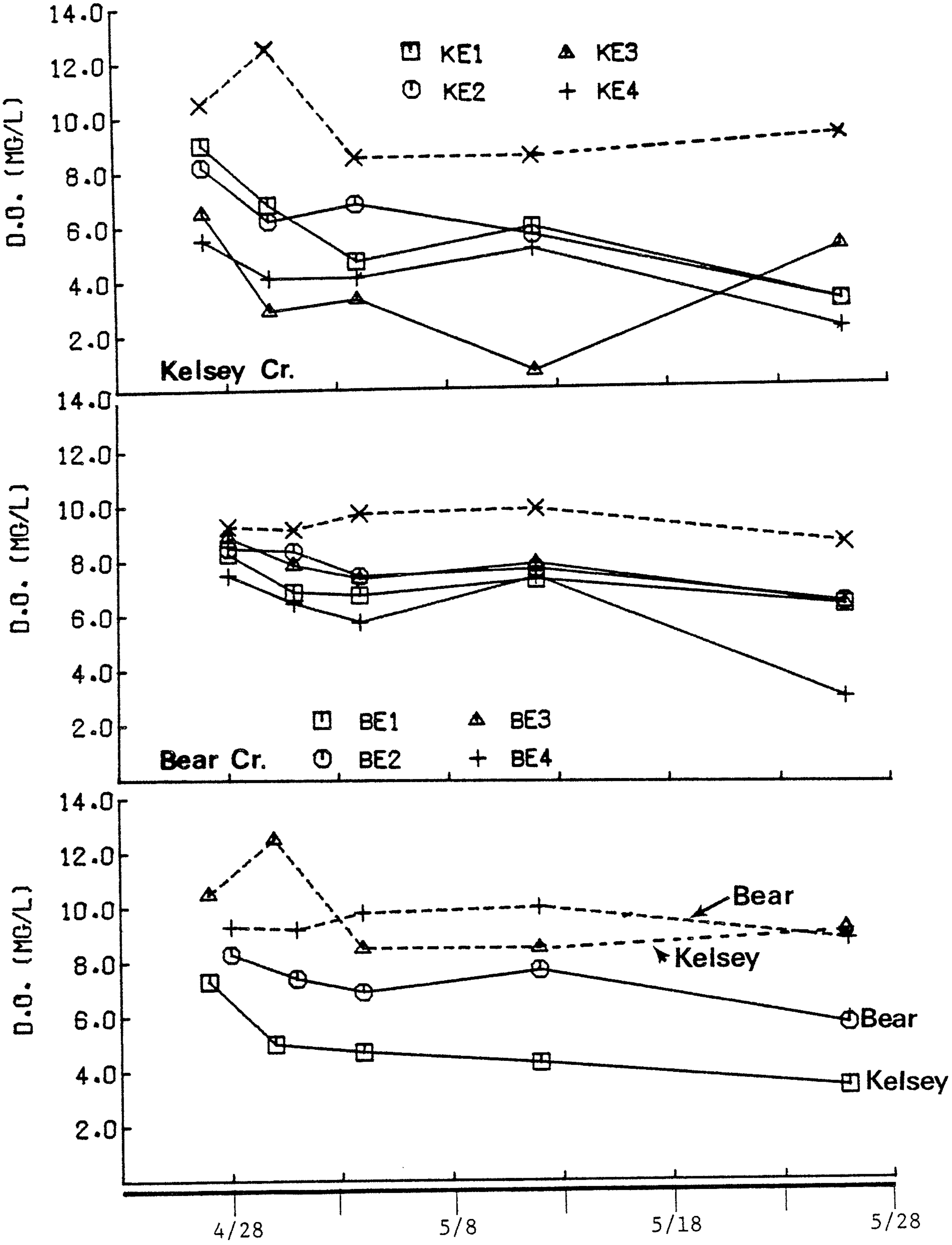


Figure 35. Mean intragravel (solid lines) and surface (dotted lines) water dissolved oxygen concentrations at sites in Kelsey Creek (upper), Bear Creek (middle), and streamwide averages (lower) during the spring instream embryo bioassays.

among them were not significant. On a stream-wide basis (Fig. 35), however, a significantly greater concentration of dissolved oxygen was present within the intragravel water of Bear Creek ($\alpha \leq .001$). Surface water dissolved oxygen concentrations (Fig. 35) were typically 85-90 percent of saturation values and differed little between streams.

Intragravel dissolved lead concentrations in both streams were insignificant, reaching a maximum of 6 μg/l at sites BE1-3 and KE1-3 (Table 22).

7.4 Salmonid Embryo Bioassays

7.4.1 Instream

The coho eggs used in the winter bioassay, which were planted immediately after water-hardening, apparently suffered extremely high mortality during the planting operations. Eggs sampled 28-38 days after initiation of the experiment had an average survival of 3.8 percent in the control stream and only 0.3 percent in Kelsey Creek. Subsequent sampling in the period from day 59 to day 75 indicated that mortality had occurred after planting as well. Average survival in Bear and Kelsey Creeks at that time was 1.5 and 0.1 percent, respectively (Fig. 36 and Table 23). Combining the sampling dates, it was found that the median survival of eggs was significantly ($\alpha \leq .05$) lower when incubation occurred in Kelsey Creek.

There is limited evidence that egg mortality, in addition to that which occurred during planting, was induced by hypoxial stress (Fig. 37). All bioassays conducted with intragravel dissolved oxgyen concentrations of 7.2 mg/l or less experienced complete egg mortality, and, in general, egg survival was greater as the mean oxygen level increased. This relationship (H_0: $B = 0$, H_A: $B > 0$) is statistically significant with $\alpha \leq .083$.

Table 22. Concentrations of soluble lead (μg Pb/l) in Kelsey and Bear Creek interstitial water sampled on April 30, 1981 (n.s. = not sampled).

		Pb concentration (μg/l)		
		Standpipe number		
Stream	Site	1	3	5
Kelsey Creek				
	KE1	<4	<4	5
	KE2	<4	6	<4
	KE3	<4	<4	<4
	KE4	5	<4	<4
Bear Creek				
	BE1	5	5	5
	BE2		not sampled	
	BE3	n.s.	<4	<4
	BE4B	n.s.	<4	<4

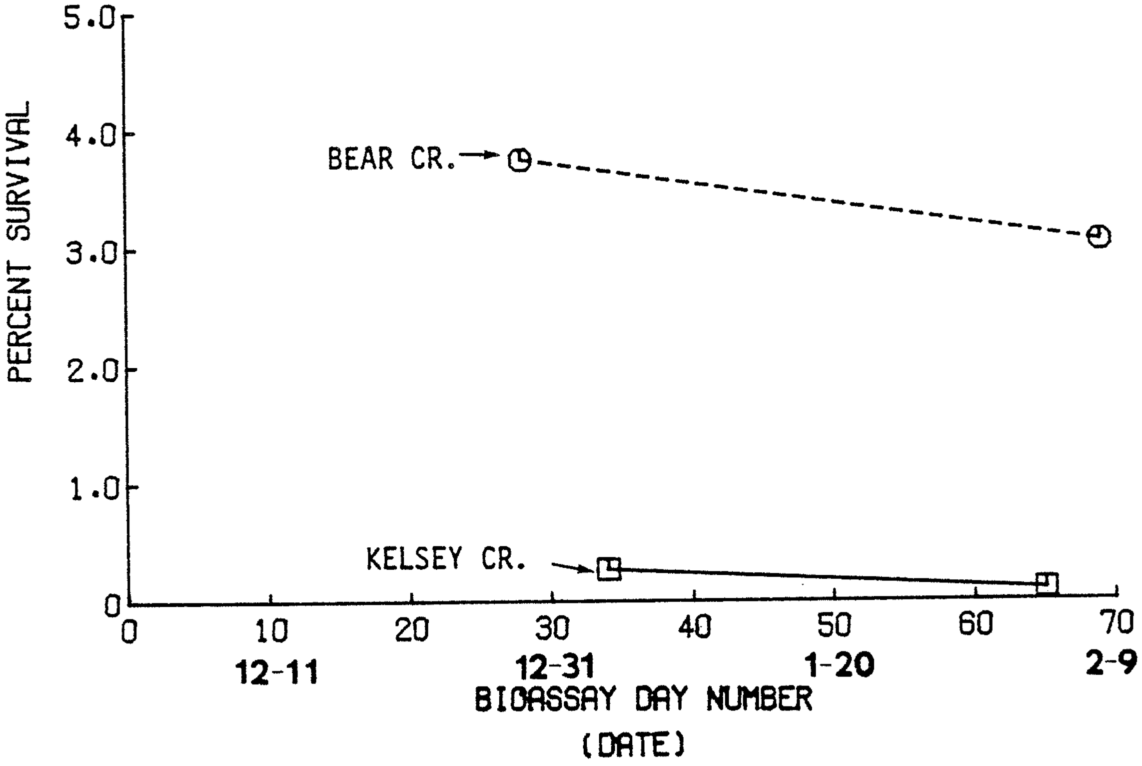


Figure 36. Estimated survival of coho salmon embryos during the winter instream bioassays.

Table 23. Estimated survival of coho salmon embryos during the winter instream embryo bioassay, by sampling date and site.

Stream	Site	Date sampled	Percent survival
Kelsey Creek		1/04/81	
	KE1-1		1.1
	KE2-3		0
	KE3-4		0
	KE4-2		0
Mean			0.27
Bear Creek		12/29/80	
	BE1-3		4.9
	BE2-1		0.9
	BE3-2		5.4
	BE4-2		3.8
Mean			3.75
Kelsey Creek		2/04/81	
	KE1-5		0
	KE2-1		0.4
	KE3-3		0
	KE4-1		0
Mean			0.10
Bear Creek		2/08/81	
	BE1-1		3.2
	BE2-3		0
	BE3-3		2.9
	BE4-1		0
Mean			3.05

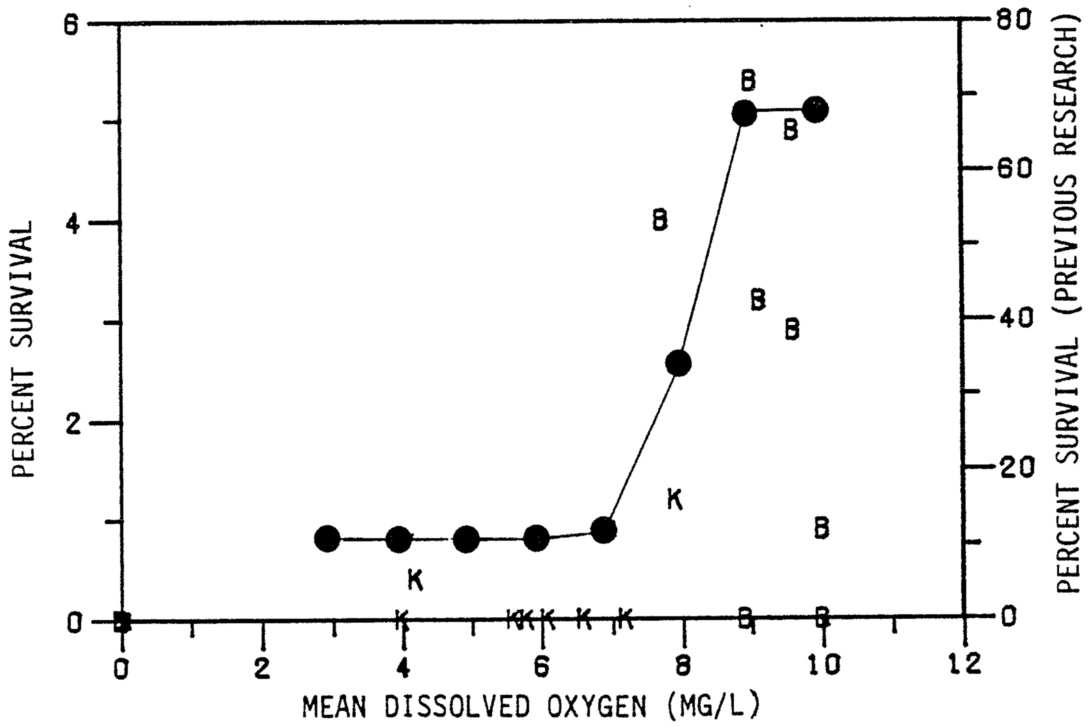


Figure 37. Relationship between the survival of coho salmon embryos and mean intragravel dissolved oxygen concentrations during the winter instream bioassays (K — Kelsey Creek; B — Bear Creek; ●——● prior research).

Due to the limited egg survival, the coho embryo bioassay was terminated after two sampling dates.

The spring instream embryo bioassay utilized eyed rainbow trout eggs. Embryos sampled in Bear Creek after incubation periods of 9, 22, and 34 days had average survivals of 30.5, 1.5, and 6.4 percent (Fig. 40 and Table 23). The survival of eggs sampled at concurrent time intervals in Kelsey Creek was 26.1, 1.3, and .2 percent (Fig. 38 and Table 24). No significant differences existed between the median survival of embryos incubated in each stream during the spring bioassay.

As might be expected, the survival of eggs up to the initial sampling date of the spring bioassay is not correlated with mean intragravel oxygen concentrations. It is likely that this relation was obscured by the high initial planting mortality. A significant regression ($\alpha \leq .001$) does exist between the survival of eggs recovered during the final two sampling dates and the mean dissolved oxygen concentration which they experienced (Fig. 39). No eggs survived during the spring bioassays when intragravel oxygen levels averaged less than 6.5 mg/l.

No relationship was found between the percent of fines in which the eggs were incubated and their survival during either the winter or spring bioassays.

7.4.2 Streamside

Dissolved oxygen concentrations in the Kelsey Creek streamside egg incubation box declined throughout the bioassay period, ranging from 11.5 mg/l to 5.3 mg/l (Table 25). Water samples for lead analysis were not taken until April 30, slightly less than two months after termination of the bioassays. Dissolved lead concentrations in both the Kelsey Creek and Valley Creek

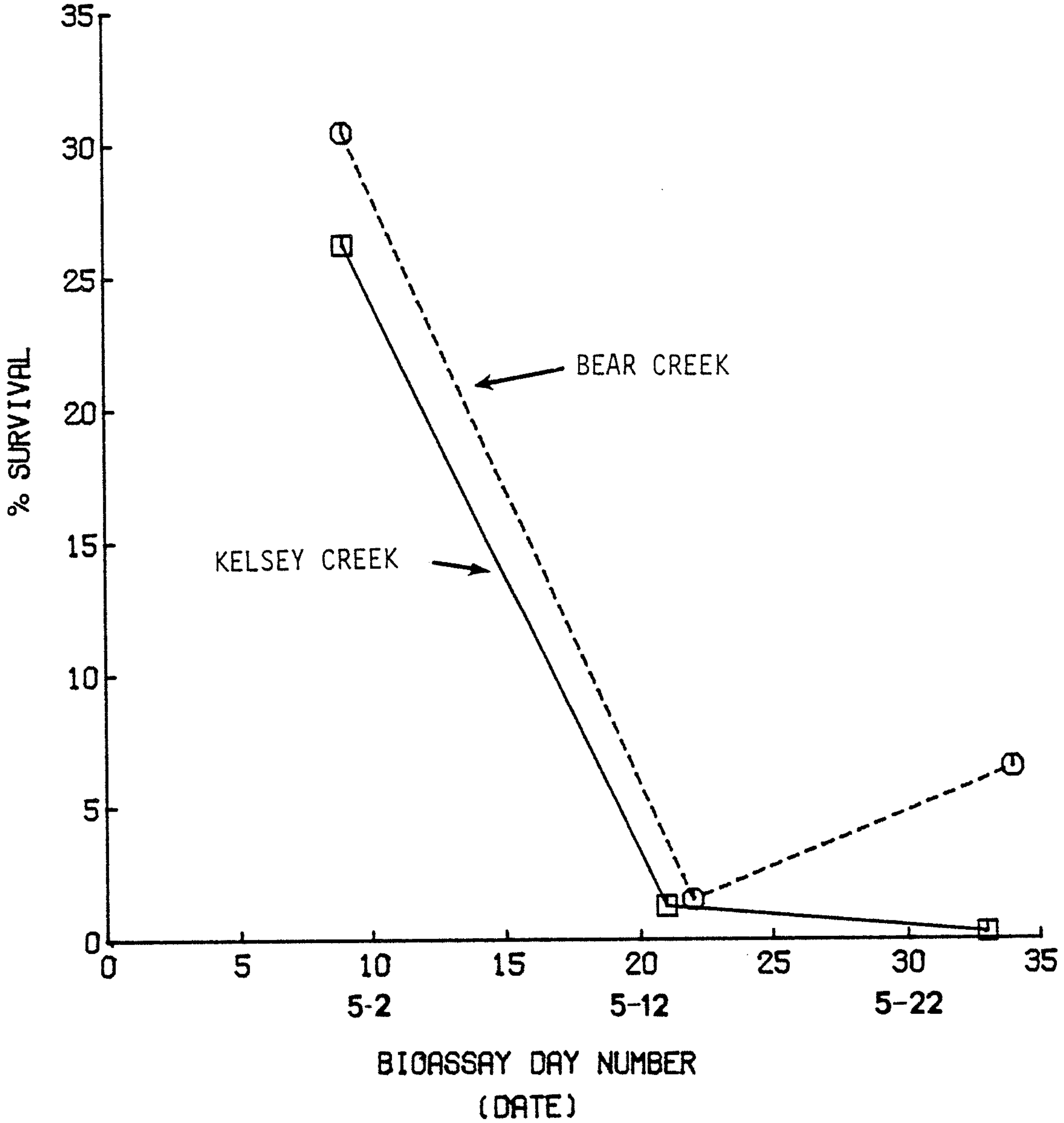


Figure 38. Estimated survival of rainbow trout embryos during the spring instream bioassays.

Table 24. Estimated survival of rainbow trout embryos in the spring instream embryo bioassays, by sampling date and site.

Stream	Site	Date sampled	Percent survival
Kelsey Cr.		5/02/81	
	KE1-1		.30
	KE2-3		.25
	KE3-1		.08
	KE4-3		.42
		Mean	.26
Bear Cr.		5/02/81	
	BE1-3		.39
	BE2-5		.34
	BE3-1		.30
	BE4-3		.19
		Mean	.31
Kelsey Cr.		5/14/81	
	KE1-3		.02
	KE2-1		.03
	KE3-5		0
	KE4-5		0
		Mean	.01
Bear Cr.		5/15/81	
	BE1-1		.02
	BE2-3		0
	BE3-5		.01
	BE4-1		.03
		Mean	.01
Kelsey Cr.		5/26/81	
	KE1-5		0
	KE2-5		.01
	KE3-3		0
	KE4-1		0
		Mean	<.01
Bear Cr.		5/27/81	
	BE1-5		.09
	BE2-1		.10
	BE3-3		.07
	BE4-5		0
		Mean	.06

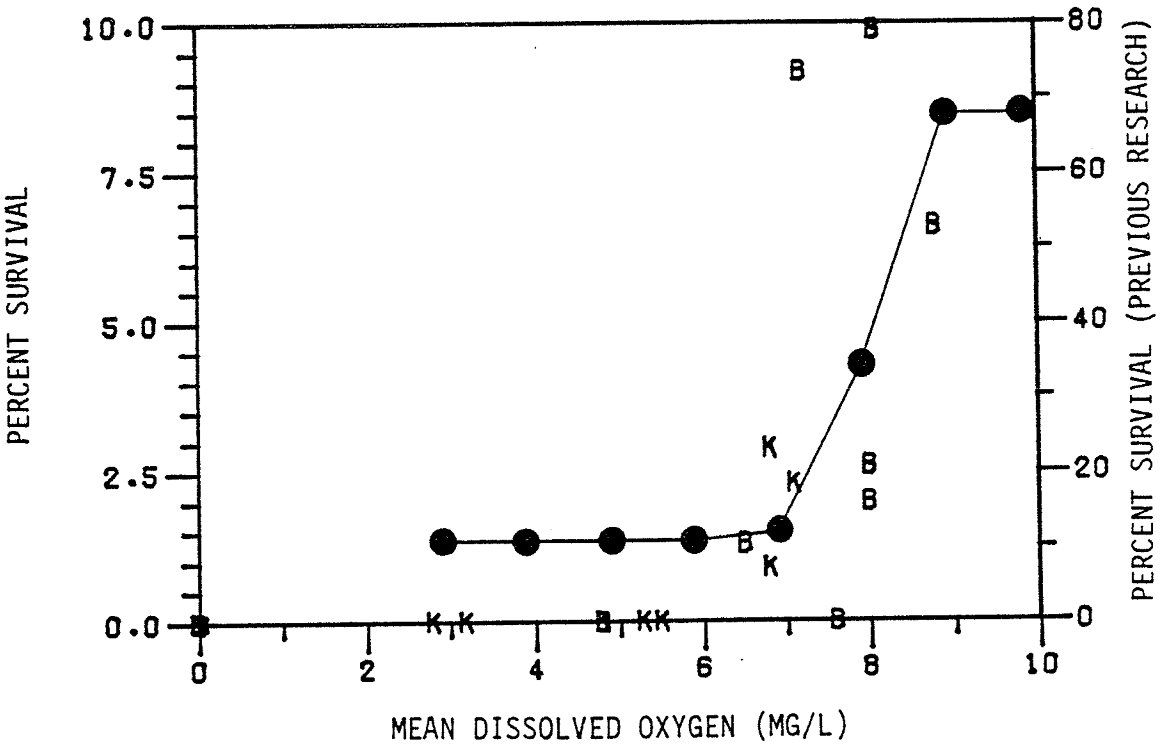


Figure 39. Relationship between the survival of rainbow trout embryos and mean intragravel dissolved oxygen concentrations during the spring instream bioassays (K — Kelsey Creek; B — Bear Creek; ●——● prior research).

Table 25. Dissolved oxygen and soluble lead concentrations in streamside bioassay water.

Date	Dissolved oxygen (mg/l)	Soluble lead (μg/l)
1/30/81	11.0	
2/05/81	11.5	
2/20/81	6.2	
2/27/81	6.9	
3/06/81	5.3	
4/30/81		<4

incubation boxes were less than 4 μg/l at that time (Table 25).

The estimation of embryo survival in the streamside bioassay was complicated by the absence of a complete complement of embryos in most of the Vibert boxes at the time of sampling. When this occurred, Vibert boxes were randomly removed for sampling until one was located which had approximately 20 embryos in it. The estimated survival of embryos was always greater than 95 percent (Table 26), regardless of whether the box sampled had missing embryos or not.

7.5 Population and Age Estimates

7.5.1 Species Composition

Kelsey and Bear Creeks were characterized by different fish assemblages, although those species present in only one of the streams were generally uncommon. The prickly sculpin (*Cottus asper*) and longnose dace (*Rhinichthys cataractae*) were the dominant non-salmonid species collected in Bear Creek, while the longnose dace and largescale sucker (*Catostomus macrocheilus*) were commonly taken in Kelsey Creek at stations KC4, KC5, KC9, and KC10. The upstream movement of these two species appears to be restricted by the culvert located at N.E. 8th Ave. Several adult peamouth (*Mylocheilus caurinus*) having the bright red coloration characteristic of spawning fish were captured at KC5 in late April of 1979.

Peamouth have been reported in Lake Washington (Scott and Crossman 1975); however, the incidence of stream spawning in this species is infrequent (Carl et al. 1967).

The adult steelhead trout (*Salmo gairdneri*) collected from KC4 on July 3, 1979, is believed to have strayed into the stream since juveniles of this species were not captured in Kelsey Creek. Two rainbow trout were caught

Table 26. Percent of sockeye salmon embryos surviving in streamside bioassays on each sampling date.

Date	Day	Lot number	Embryos Live	Embryos Dead	Percent survival
2/02/81	7	74	19	0	100.0
		83	11	0	100.0
2/13/81	18	78	12	0	100.0
		85	4	1	95.0
2/20/81	25	82	18	1	95.0
		80, 81, 86	7	0	100.0
2/27/81	32	79	19	1	95.0
		76, 77, 84	6	0	100.0
3/06/81	39	75	20	0	100.0
		72	19	1	95.0
3/12/81	45	71	19	0	100.0
		73	3	0	100.0

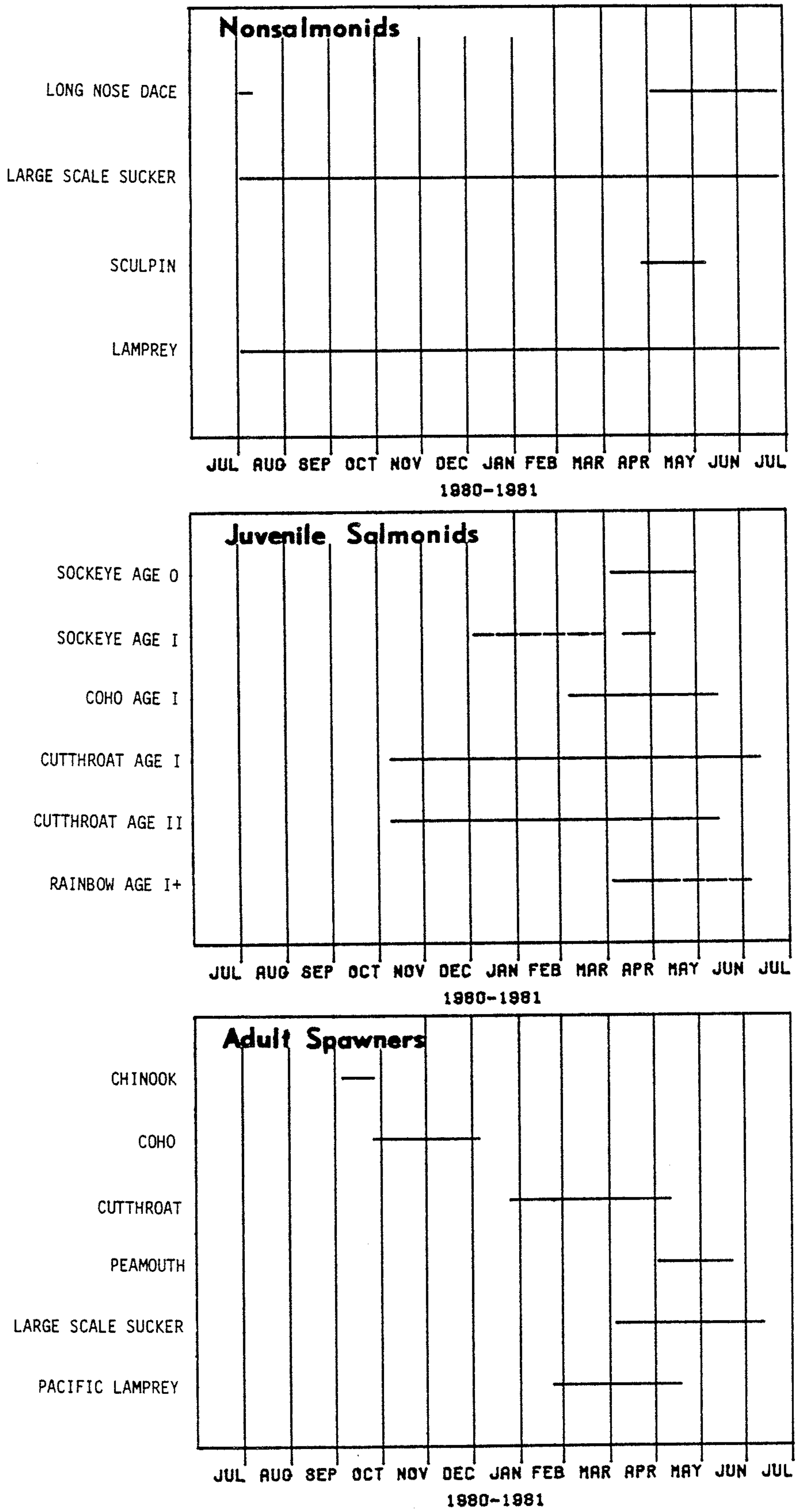
at BC3 on one occasion; the similar size of the fish (151 and 152 mm) and the date of their capture (July 23, 1979) indicate that they were released into Bear Creek by Washington Department of Game personnel. A single chinook salmon (Oncorhynchus tshawytscha) juvenile was taken in the same BC3 sample. The abundance of this species may be underestimated in census data from both streams since a few chinook adults were observed in spawner surveys conducted in 1979 and 1980.

7.5.2 Temporal Distribution and Number of Outmigrants

The seasonal variation in the composition of net catches is shown in Figure 40 for the 13 most common species-age groups. Only two species, juvenile large-scale sucker (Catostomus macrocheilus) and lamprey ammocoetes (Lampetra richardsoni and Entosphenus tridentatus) were present throughout the entire sampling period. Other non-salmonids captured by the net were longnose dace (Rhinichthys cataractae), adult peamouth (Mylocheilus caurinus), adult large-scale sucker, adult Pacific lamprey, sculpins (Cottus sp.), brown bullhead (Ictalurus nebulosus), bluegill (Lepomis macrochirus), threespine stickleback (Gasterosteus aculeatus), and smallmouth bass (Micropterus dolomieui).

Salmonid smolts typically were caught from mid-March to July (Fig. 40), although limited number of cutthroat began to appear in the net catches as early as November 4. Other species of salmonids sampled included age I coho salmon, age 0 and I sockeye salmon (released from the streamside incubation boxes), age 0 and I+ chinook salmon, and age I+ rainbow trout. Adult cutthroat trout were also frequently captured.

Coho smolts were the most prevalent of the four naturally spawned species of salmonids sampled. After the initial appearance of coho in catches on



Nonsalmonids

Juvenile Salmonids

Adult Spawners

Figure 40. Seasonal variation in the composition of the downstream net catches at Kelsey Creek.

March 9, their abundance increased rapidly until April 27, when 127 were captured (Table 27). A total of 5,753 coho smolts was estimated to have migrated from the Kelsey Creek watershed in the spring of 1981 (Table 27). Over 50 percent of this total outmigrated in the period from April 19 to May 2, and 96 percent of the smolts had left Kelsey Creek by May 16 (Fig. 42). No coho were captured after June 17.

During the three-month period of outmigration, the mean length of coho smolts did not vary significantly (Fig. 41 and Table 28). The average length of all coho measured was 109.1 mm.

The 1979 and 1980 year classes (Age I and II, respectively, in the spring of 1981) were, with the exception of two age III fish, the only age groups of cutthroat smolts captured by the net. Cutthroat of both year classes (1980 and 1979) began migrating out of Kelsey Creek as early as November 4, but smolts did not appear in appreciable numbers until near the end of March (Table 29). The maximum catch of age I cutthroat, 56 fish, occurred on April 15, and age II fish were most abundant on April 21, when 31 were captured.

The mean length of the 1980 year class of cutthroat increased during the November to July period in which they were present in net catches (Fig. 43 and Table 28). Fish captured prior to March 22 were significantly ($\alpha \leq .10$) smaller than those which migrated at a later date. The average length of all age I outmigrants measured was 134.1 mm.

The length distribution of the 1979 cutthroat year class also varied during the sampling period (Fig. 43 and Table 28). Although this trend was somewhat erratic, the largest fish were generally captured during the peak of outmigration, with smaller fish being more abundant before and after this time. The mean length of all age II cutthroat measured was 174.9 mm.

Table 27. Catch of age I coho salmon outmigrants by period, the estimated number of outmigrants, and its associated standard deviation.

Period	Dates	Days fished	Catch	$\hat{N}$	$S(\hat{N})$
5	2/22-3/21/81	4	4	85	45
6	3/22-4/04/81	6	45	318	70
7	4/05-4/18/81	12	152	537	98
8	4/19-5/02/81	7	519	3145	532
9	5/03-5/16/81	9	194	914	163
10	5/17-5/30/81	3	48	679	148
11	5/31-6/13/81	11	10	39	14
12	6/14-6/27/81	6	5	35	17
Totals		58	977	5753	590

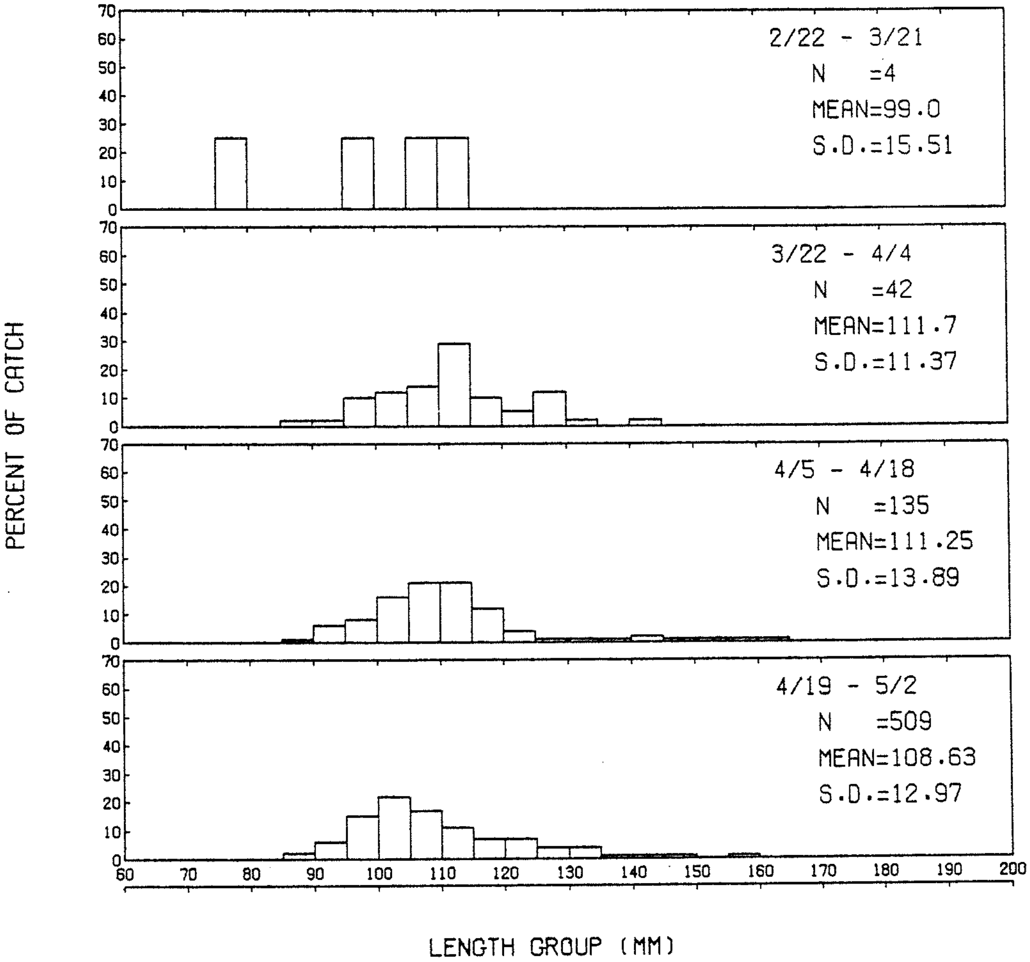


Figure 41. Normalized length frequency histograms for coho salmon captured in the Kelsey Creek downstream migrant net.

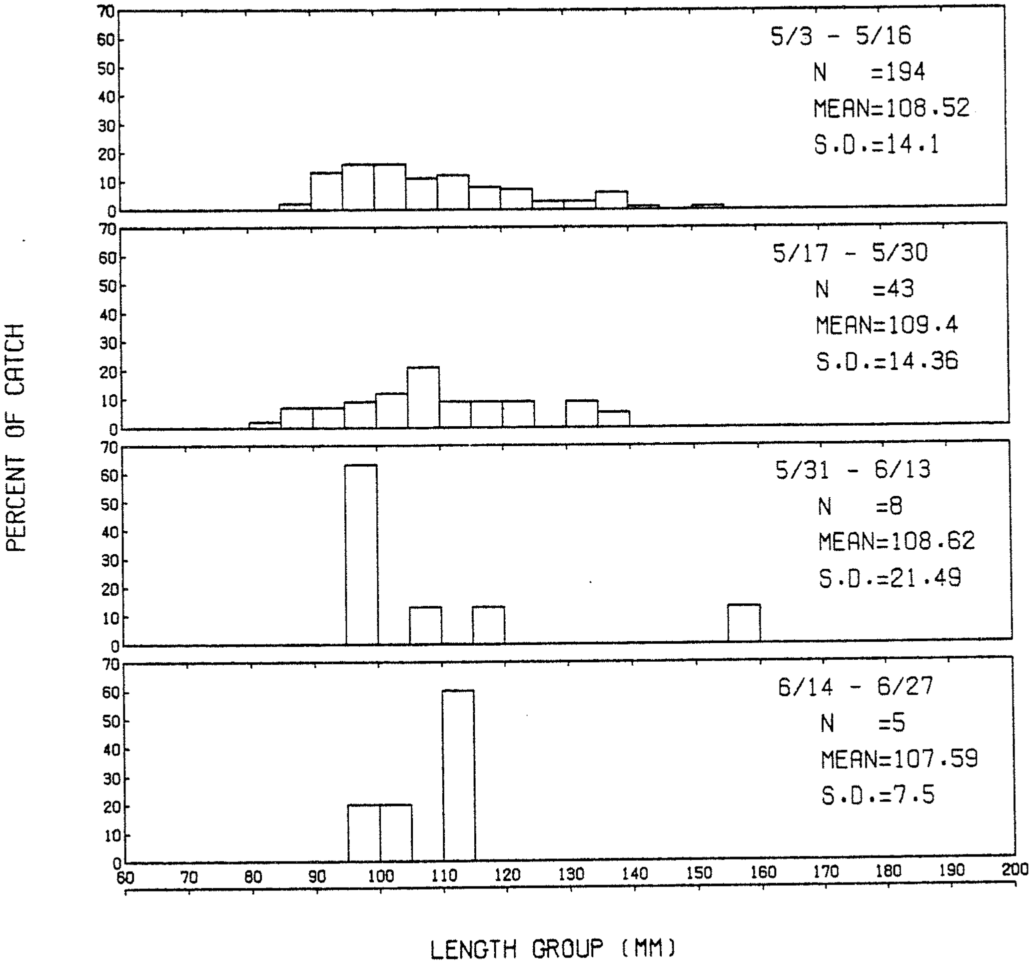


Figure 41. Normalized length frequency histograms for coho salmon captured in the Kelsey Creek downstream migrant net (continued).

Table 28. Sample size, mean length, and standard deviation of mean length of age I cutthroat trout, age II cutthroat trout, and age I coho salmon captured at the net, by sampling period.

		Cutthroat						Coho		
		Age I			Age II			Age I		
Period	Dates	n	$\bar{L}$(mm)	$S_{\bar{L}}$	n	$\bar{L}$(mm)	$S_{\bar{L}}$	n	$\bar{L}$(mm)	$S_{\bar{L}}$
1	11/02/80-11/29/80	5	102.2	13.4	5	158.0	9.3	0		
2	11/30/80-12/27/80	1	71.0	—	5	170.0	10.8	0		
3	12/28/80-01/24/81	1	131.0	—	0			0		
4	1/25/81-02/21/81	10	103.7	7.5	7	174.1	9.4	0		
5	2/22/81-03/21/81	10	103.9	4.3	13	154.5	6.1	4	99.0	7.8
6	3/22/81-04/04/81	27	141.0	2.8	47	180.6	2.7	42	111.1	1.8
7	4/05/81-04/18/81	148	141.4	1.2	79	182.4	1.5	135	111.3	1.2
8	4/19/81-05/02/81	103	130.5	1.6	56	171.0	2.0	509	108.6	0.6
9	5/03/81-05/16/81	66	133.0	1.8	18	160.7	3.2	194	108.5	1.0
10	5/17/81-05/30/81	4	124.7	6.0	1	187.0	—	43	109.0	2.2
11	5/31/81-06/13/81	33	133.2	2.5	2	156.0	12.0	8	108.6	7.6
12	6/14/81-06/27/81	14	137.1	3.9	2	194.5	22.5	5	107.6	3.2
13	6/28/81-07/11/81	2	123.5	9.5	0			0		
Mean			134.1			174.9			109.1	

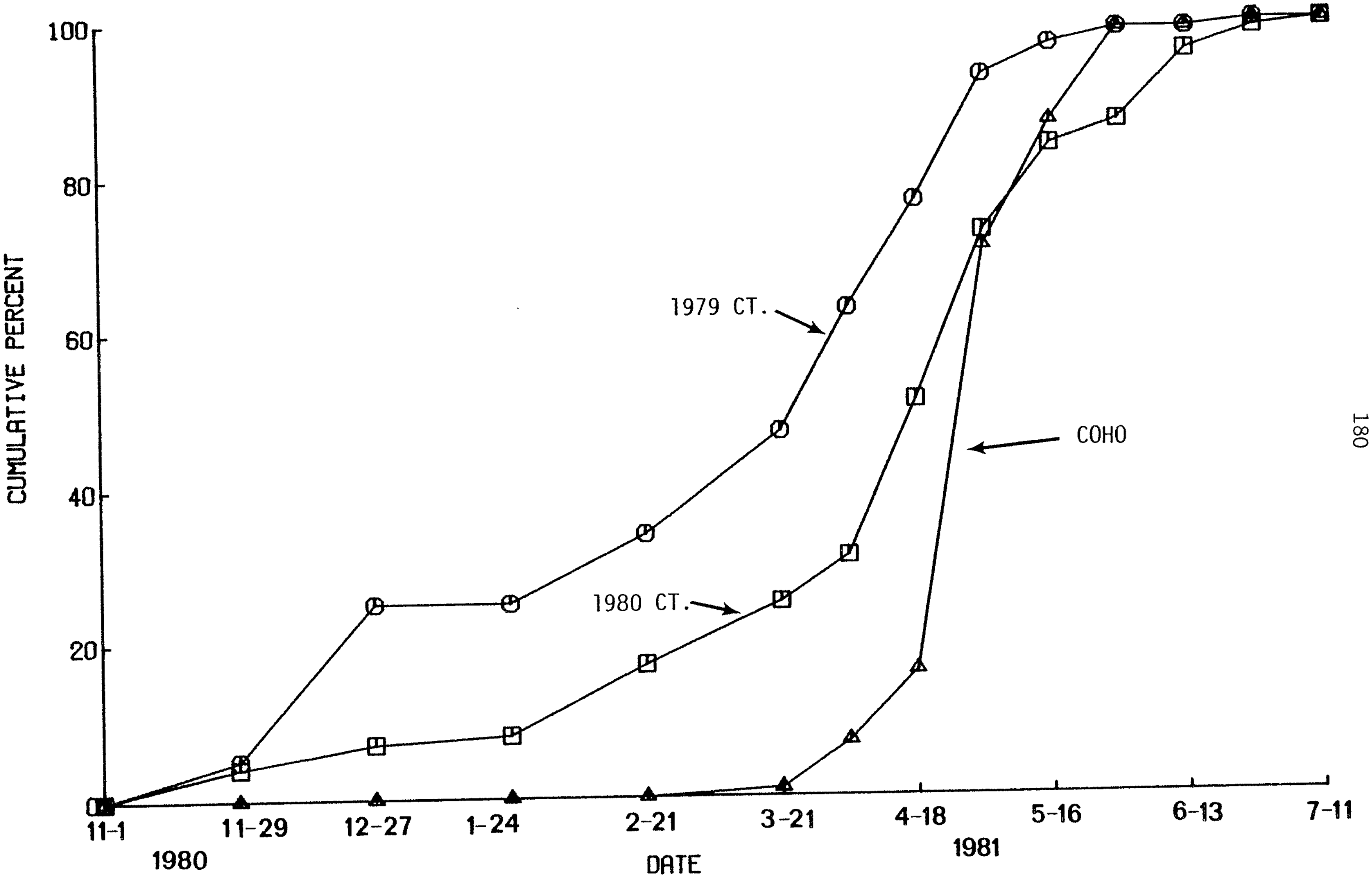


Figure 42. Cumulative percentage of age I cutthroat trout (1980 year class), age II cutthroat trout (1979 year class), and coho salmon estimated to have emigrated from Kelsey Creek in 1980-81.

Table 29. Catch of cutthroat trout outmigrants by age class and period, the estimated number of outmigrants, and its associated standard deviation.

Period	Dates	Days fished	Age I			Age II			Age III		
			Catch	$\hat{N}$	$S(\hat{N})$	Catch	$\hat{N}$	$S(\hat{N})$	Catch	$\hat{N}$	$S(\hat{N})$
1	11/02/80-11/29/80	4	5	106	90	5	106	90	0		
2	11/30/80-12/27/80	1	1	85	48	5	424	202	0		
3	12/28/80-01/24/81	3	1	28	29	0			0		
4	1/25/81-02/21/81	3	10	283	101	7	198	82	4	57	30
5	2/22/81-03/21/81	4	10	212	75	13	276	89	0		
6	3/22/81-04/04/81	6	27	191	48	47	332	73	0		
7	4/05/81-04/18/81	12	166	587	84	87	308	60	0		
8	4/19/81-05/02/81	7	104	630	120	56	339	72	0		
9	5/03/81-05/16/81	9	66	311	64	18	85	24	0		
10	5/17/81-05/30/81	3	6	85	37	3	42	25	0		
11	5/31/81-06/13/81	11	67	258	53	2	8	6	0		
12	6/14/81-06/27/81	6	14	99	31	2	14	10	0		
13	6/28/81-07/11/81	3	2	28	25	0			0		
Totals		72	479	2904	246	245	2132	281	4	57	30

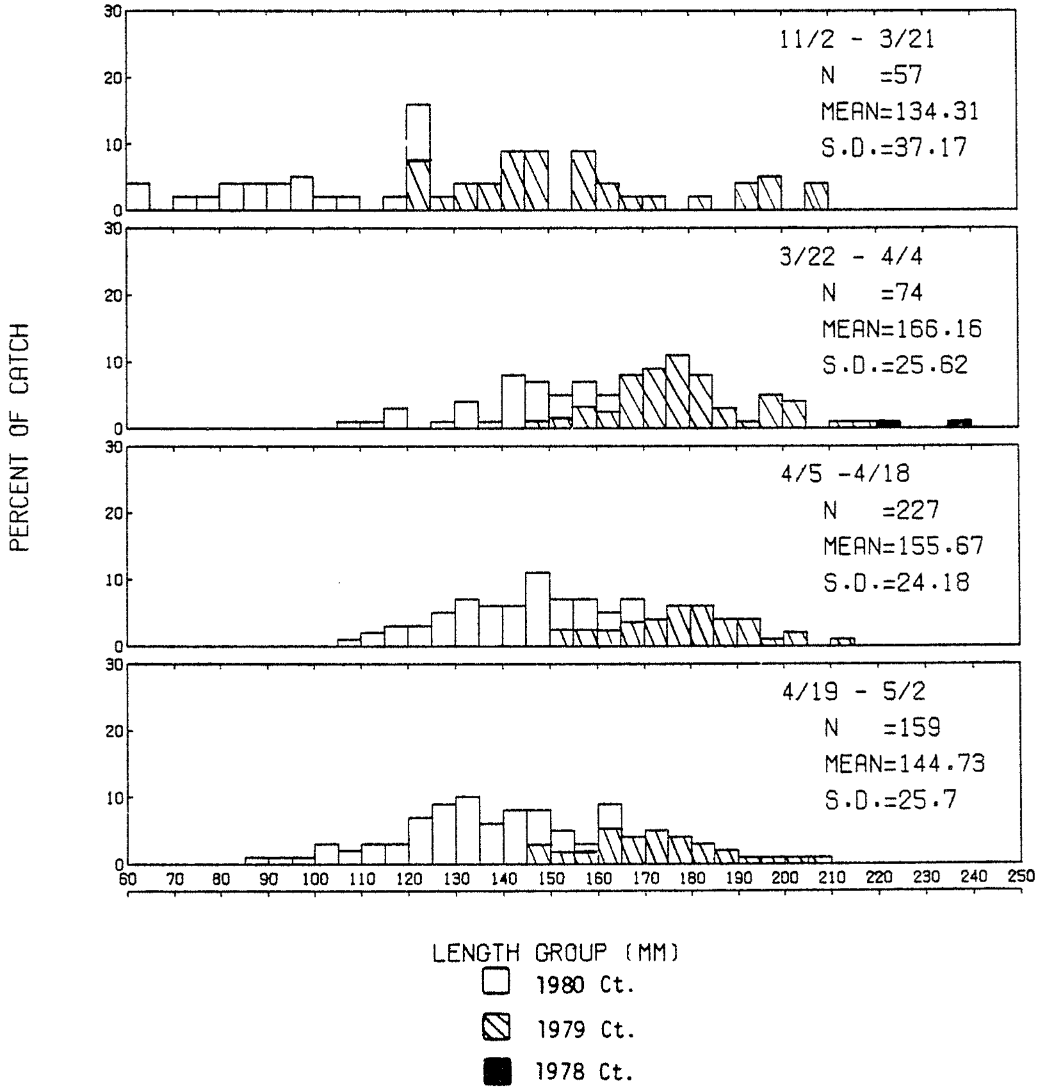


Figure 43. Normalized length frequency histograms for cutthroat trout captured in the Kelsey Creek downstream migrant net.

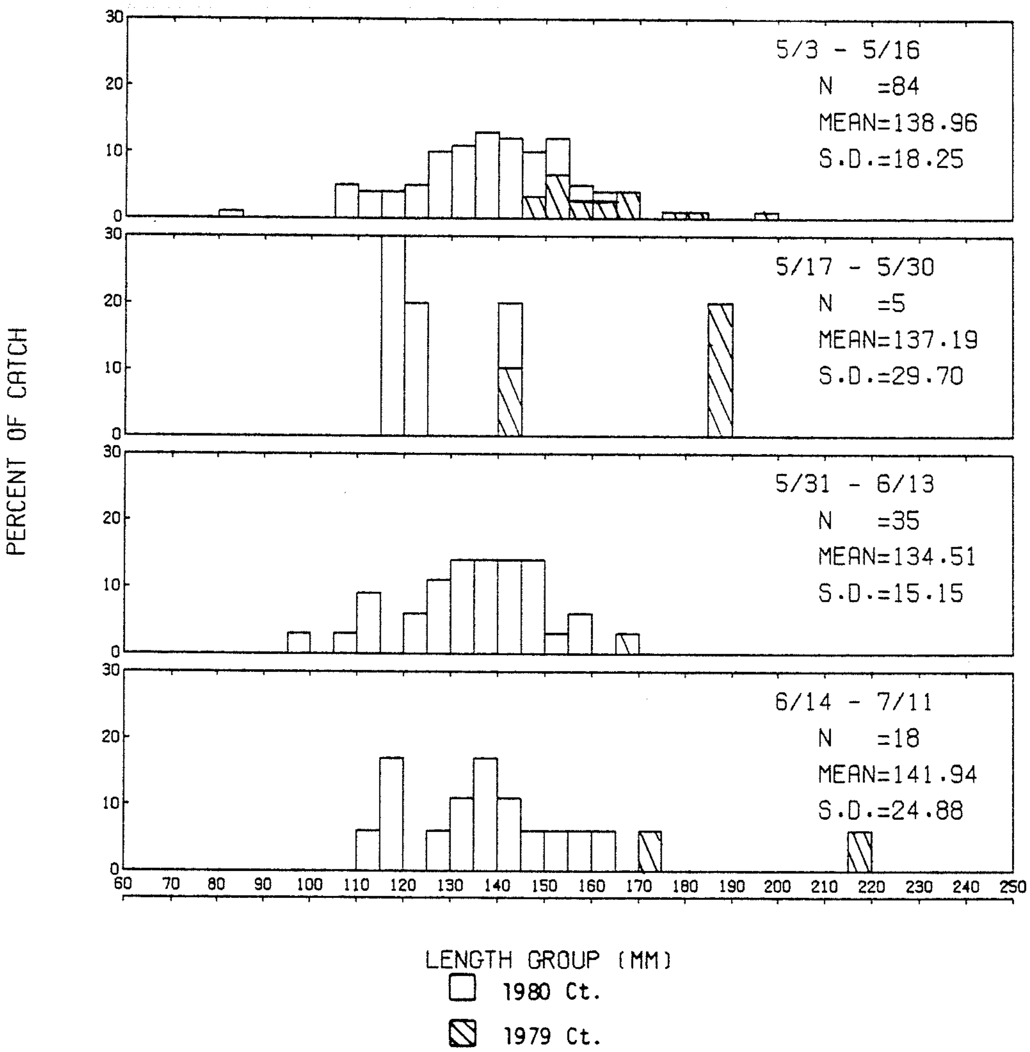


Figure 43. Normalized length frequency histograms for cutthroat trout captured in the Kelsey Creek downstream migrant net (continued).

A total of 5,093 cutthroat trout were estimated to have migrated from the Kelsey Creek watershed during 1980-81. The majority of these (57.0 percent) were age I fish, with age II fish contributing the bulk of the remainder (41.9 percent). Age II cutthroat displayed a tendency to migrate from Kelsey Creek at an earlier date than those of age I, which did not begin to dominate catches until early April (Table 29 and Fig. 43). For this reason, the date at which 96 percent of the age I smolts had migrated from the creek was approximately one month later for age II fish (June 13 and May 16, respectively).

The only flood of even moderate magnitude for which net data are available occurred on November 26-27, 1980, when the peak discharge (during net operation) reached 4.27 m^3/sec. Catches of juvenile cutthroat, juvenile large-scale sucker, and lamprey at this time were all slightly greater than during the previous two nonflood sampling dates (Table 30). The relative abundance of cutthroat in the catch on November 26 may be a reflection of seasonal variation rather than flood effects, as an even greater number were captured on the following sampling date (Table 30).

The net was fished during two significant drought-ending rainstorms in the fall of 1980 (August 17 and October 13), and during a nonsignificant drought-ending rainstorm in the summer of 1981 (July 29) (Table 31). The storm which ended the 33-day drought in August of 1980 was of considerable intensity, and the discharge in Kelsey Creek increased by 2.12 m^3/sec over base flow. Catches of juvenile large-scale sucker, bluegill, lamprey ammocoetes, and stickleback were all considerably greater during this storm than during the previous two dates on which sampling was conducted. The numbers and species of fish captured in the net during storms which ended the latter two drought

Table 30. The impact of a minor flood event (11/26/80) upon catches at the downstream net. Species abbreviations: Ct - cutthroat trout, LSS - largescale sucker, BG - bluegill, Am - lamprey ammocoete, Stb - stickleback.

Date	Q_{max}[1] (m^3/s)	Prior/post sampling dates	Time fished	Catch				
				Ct	LSS	BG	Am	Stb
11/26-27/80	4.3		15:30-11:30	4	13	0	18	1
	0.3	11/16-17/80	15:30-08:00	1	0	0	2	0
	0.7	11/25-26/80	12:00-08:30	2	7	1	2	0
	—[2]	12/13-14/80	15:00-08:00	6	3	1	2	3

[1] From U.S.G.S. gauging records.

[2] U.S.G.S. gauge inoperative.

Table 31. The impact of drought-ending rainstorms upon the catch of fish at the downstream net. Species abbreviations: Rb — rainbow trout, Ct — cutthroat trout, So — age 0 sockeye salmon, LND — long-nose dace, LSS — largescale sucker, BG — bluegill, Stb — stickleback, Am — lamprey ammocoetes.

Date	Rise in Q (m^3/s)[1]	Previous two sampling dates	Time fished	$\bar{Q}^1$ (m^3/s)	Days from previous rain	Catch							
						Rb	Ct	So	LND	LSS	BG	Stb	Am
8/17-18/80	2.1		16:15-06:30	1.1	33[3]					106	9	2	22
		8/03-04/80	17:00-06:30	[2]					1	26			
		8/10-11/80	18:15-19:45	[2]					2	11			1
10/24-25/80	0.3		15:30-07:30	0.4	16[3]					4			2
		10/08-09/80	18:30-08:00	0.2						5			
		10/18-19/80	18:00-09:30	0.2									1
7/28-29/81	0.2		17:00-17:00	0.2	16[4]								1
		7/10-11/81	17:00-17:00	0.3		1	3	4	2	3		11	8
		7/20-21/81	17:00-17:00	0.2			1	1		1			2

[1] From U.S.G.S. gauging records.

[2] U.S.G.S. gauge inoperative.

[3] Pitt et al. (1981)

[4] National Weather Service, Sea-Tac Station.

periods (October 13 and July 29) did not appear to differ qualitatively from those of the two sampling days which preceded them.

7.5.3 Population Estimates

The data relating to cutthroat trout and coho salmon population estimates for Bear and Kelsey Creek stations on each of the sampling dates are summarized in Appendix Tables 4 and 5. In those instances where only a few individuals were caught in early passes, the actual number of fish caught rather than the number estimated by the removal method was assumed to represent the true population size. The accuracy of the removal method as applied to electrofishing appeared to be good under the sampling conditions. Estimates of the proportion of the total population caught in the study areas average 93 percent, ranging from 64 to 100 percent. The assumption of equal probability of capture between passes was violated in only 2 of the 91 cases for which a chi-square statistic was calculated.

Evidence for species selectivity in the electrofishing technique was not found, although it seems probable that juvenile coho are more susceptible to capture than cutthroat, since the latter population consists of a mixture of several size groups. A test of significance for differences between the mean probability of capture, $\hat{p}$, calculated for the two species rejected the hypothesis of species selectivity (t-test, $\alpha = 0.05$). There was also no detectable difference between the mean $\hat{p}$ values recorded for the separate streams, implying that the probability of capture was not affected by environmental factors such as differences in water depth, velocity, and instream cover between Kelsey and Bear Creeks.

Table 32 shows those instances where a significant correlation between

Table 32. The mean length (mm) of fish per pass in samples where a significant correlation between mean length and pass number was observed.

Station	Date	Species	Length/Pass						Probability
			1	2	3	4	5	6	
			Decreasing length with pass						
KC2	5/01/79	Cutthroat	131	115	109	115	101		.05
KC4	7/03/79	Cutthroat	120	120	116	101	107	101	.01
KC4	7/03/79	Cutthroat	68	65	61	48			.05
BC2	7/23/79	Cutthroat	112	93	76				.01
KC7	8/23/79	Cutthroat	74	71	68				.01
KC4	9/18/79	Cutthroat	83	79	73				.05
KC2	7/17/80	Cutthroat	63	54	46				.05
			Increasing length with pass						
KC4	7/25/79	Cutthroat	64	65	65	67			.05
BC3	5/06/80	Coho	109	119	125				.05

fish length and pass number was observed. Of the 64 samples for which a regression line was calculated, 7 indicated a significant decrease in mean length over successive passes, while in 2 cases an increase in fish length was observed. Clearly there is some variability associated with small sample size; however, a general tendency for larger fish to be caught in early passes is supported by the data. For example, the number of age 0 cutthroat in Kelsey Creek increases steadily in the samples following fry emergence in July, reaching a peak in the August samples. Below a certain size, small fish appeared to be less sensitive to the potential gradient of the electroshocker which implies that the actual number of age 0 fish present in Kelsey Creek in July was underestimated. Further support of size selectivity is provided by the observation that the mean length of fish caught in the last pass was less than the mean length of fish caught in the penultimate pass in 62 percent of the samples. A significant relation between length and pass number probably would have been detected had a greater number of passes been made.

Although unequal catchability among age groups is suggested, the relatively high estimates of the fraction of the total population caught indicate that the data will not result in serious overestimates of the mean length, weight or age of uncollected fish.

7.5.3.1 Population Density and Regulatory Mechanisms

Seasonal changes in the density of cutthroat and coho age groups in the period April 1979 to July 1980 are given in Figures 44-46 with survivorship curves drawn where a significant correlation coefficient was found for the regression of log density against time. A significant trend was detectable in those age groups present in sufficient numbers where sampling variability

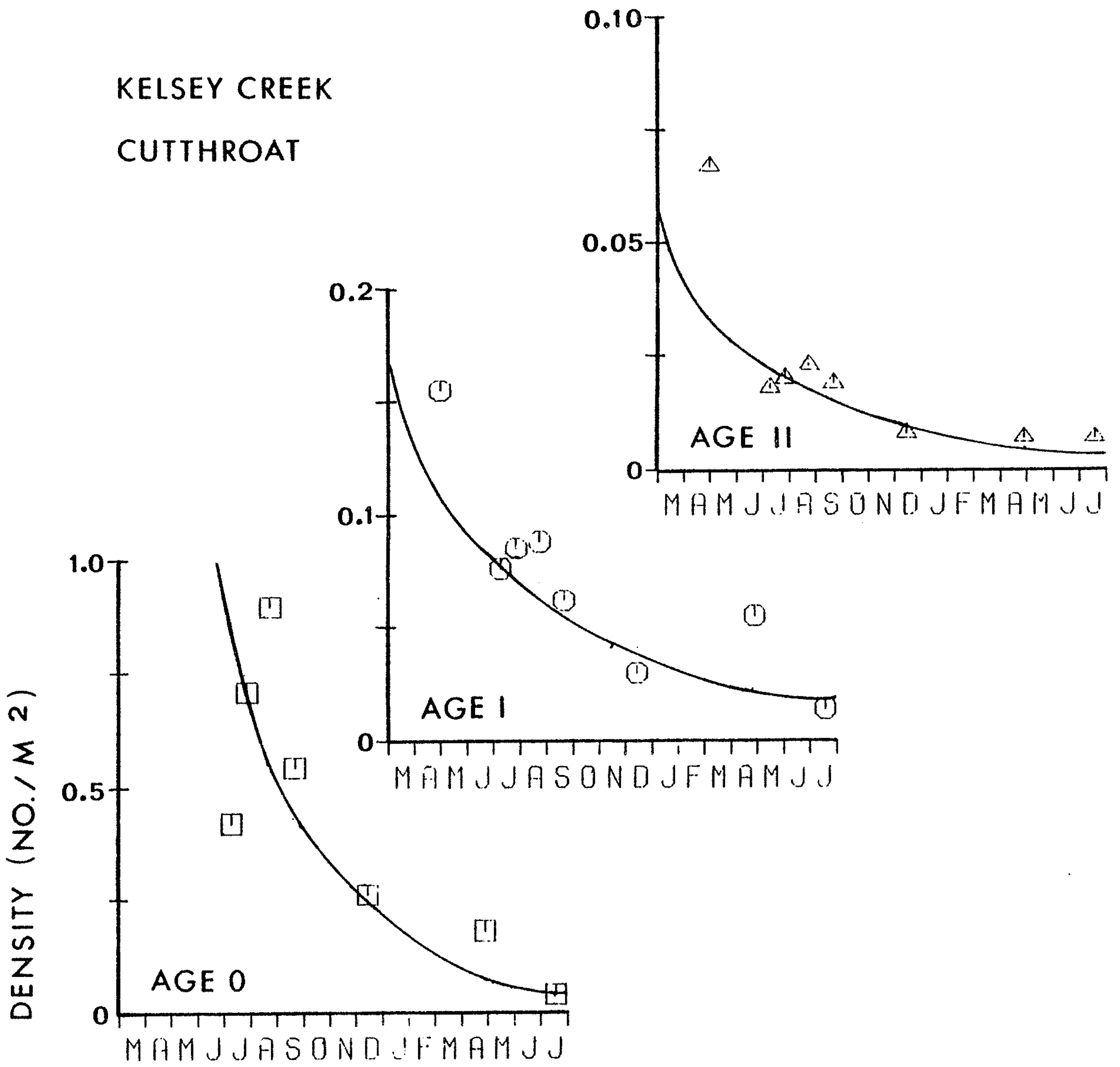


Figure 44. The average density of the 1977 (age II), 1978 (age I), and 1979 (age 0) year classes of cutthroat trout at the Kelsey Creek sampling sites in the period April 1978 to July 1979.

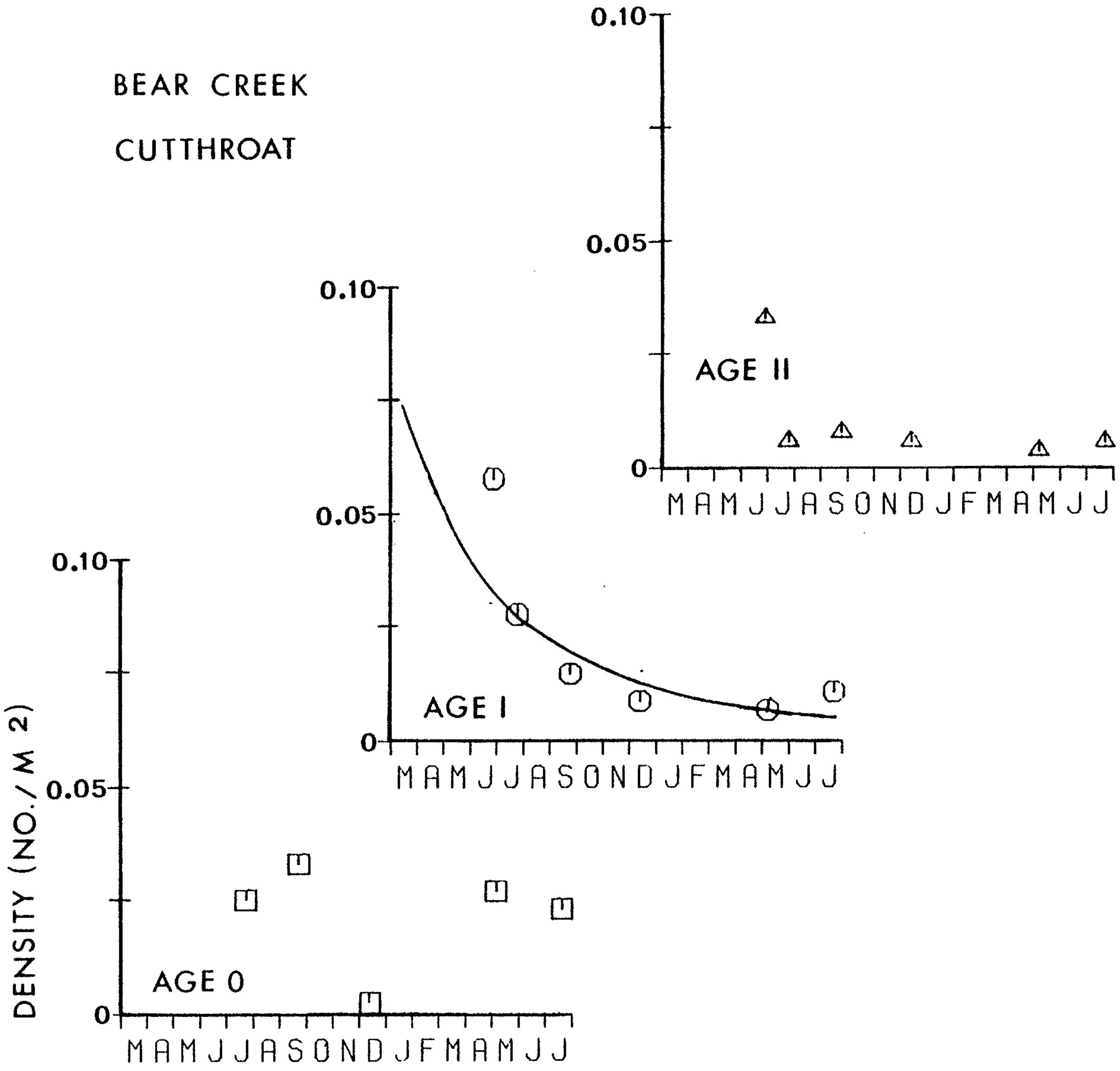


Figure 45. The average density of the 1977 (age II), 1978 (age I), and 1979 (age 0) year classes of cutthroat trout at the Bear Creek sampling sites in the period May 1978 to July 1979.

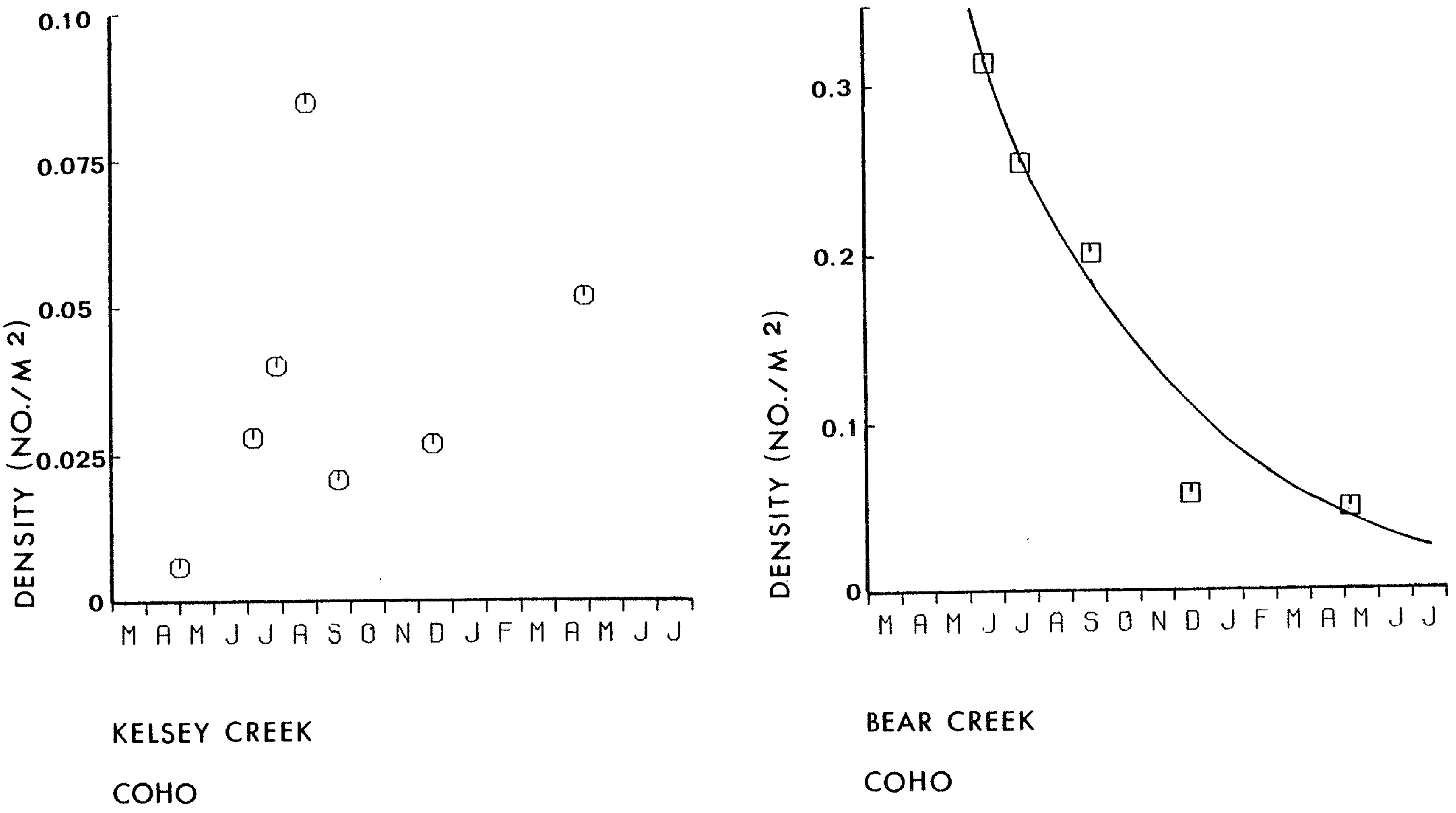


Figure 46. The average density of the 1979 year class of coho salmon at sampling sites in Kelsey and Bear Creeks in the period April 1978 to July 1979.

and migratory behavior did not obscure the expected decline in population size. There is some indication that the density of coho salmon in Kelsey Creek varied in response to the contribution of fish from tributary streams, a point which will be discussed further in relation to biomass estimates. Age III+ cutthroat (1976+ year class) were not abundant enough in either stream to plot survivorship curves.

After emergence in late May and June 1979, the density of underyearling cutthroat trout in the Kelsey Creek samples increased to a maximum of 0.895 fish/m^2, a conservative estimate because electrofishing for very small trout was inefficient. For this reason the density estimates for July 8 and 26 of 1979 were not used to calculate the survivorship curve for age 0 trout. The actual density of this cohort on the two July sampling dates is probably closer to the estimates of 0.86 and 0.74 fish/m^2, respectively, calculated from the regression equation.

From a comparison of the 1979 year class density of cutthroat in Bear Creek with the 1978 and 1979 year classes, it became apparent that the recruitment of young-of-the-year trout in 1979 was very low. Densities of the total salmonid population in Kelsey Creek ranged from 1.091 fish/m^2 in August 1979, of which 82 percent were age 0 trout, to 0.238 fish/m^2 in July 1980, with 74 percent of the samples comprised of age 0 trout. The relative abundance of the 1979 cutthroat year class in July 1979 was approximately 4 times as great as the 1980 year class in July 1980. In Bear Creek, total salmonid densities ranged from 0.091 fish/m^2 in December 1979 to 0.518 fish/m^2 in May 1980 shortly after the emergence of coho fry. In spite of coho mortality incurred during the interval of May to July 1980, the recruitment of cutthroat fry in the Bear Creek study areas resulted in high densities (0.515) measured on the latter date. The density of Bear Creek coho and cutthroat juveniles in the December samples was conspicuously low, suggesting an active migration out of the study areas, possibly into overwintering habitat in low gradient areas of the stream. Such winter behavior would be a means of avoiding unprofitable energy expenditure. Additionally, the low density estimates may reflect sampling inefficiency under winter conditions of high discharge and low water temperature.

The size of the 1979 coho year class in the Bear Creek samples declined from a maximum of 0.318 fish/m^2 in June to 0.057 fish/m^2 in May of the following year. The latter value may be considered a somewhat inflated estimate of smolt yield from the stream since pre-smolt mortality among the remaining fish was probably greater than the number of fish which had already emigrated downstream.

Large fluctuations in Kelsey Creek coho densities are due in part to the variability associated with sampling small populations. The number of fish collected tended to increase as the summer progressed as was observed for age 0 cutthroat; however, the maximum density observed in August probably represented something more than full recruitment to the sample population. Stations KC6 through KC10 apparently sustained greater densities of coho than did stations KC1 through KC5. The lowermost study area on Kelsey Creek, KC5, was selected because it was the only locality with a steep gradient and diverse habitat features situated below an extensive marsh on the lower reach of the stream. Only 8 of the 167 coho sampled in stations KC1-KC5 during the study period were taken at KC5. To a lesser extent, the age structure of the cutthroat population at the same station was biased toward older fish. Consequently, the total salmonid density for a given date at KC5 was usually less than estimates for other stations on the stream.

7.5.3.2 Parameters Estimated from Mark-Recapture Experiments

Marking of fish, which commenced in July of 1980, allowed estimation of two parameters of interest, the rates of loss and replacement of fish within the study sites. The number of immigrants and the survival of marked fish between sample dates are summarized in Appendix Tables 6-14.

7.5.3.2.1 Rate of Loss

The rate of loss (L) of marked cutthroat trout from each study site is shown in Figures 47 and 48. Several cohorts have been plotted on the same graph so that mortality throughout the life of a given cohort may be simulated. Sites within a stream were generally similar in the magnitude and temporal trend of L. One notable exception occurred in the 1980 year class at KC4 during the period November to December 1980. The maximum L computed for cutthroat in Kelsey Creek, .0214/day, occurred at KC4 in the period May to July 1981. The maximum rate observed in Bear Creek was .0166/day.

Comparisons among streams may be most easily made by computing the mean L for all sites on a given stream at each date. For Kelsey Creek, the data from KC4 for the period November to December has not been included because of its aberrant nature. The trend in the rate at which cutthroat disappear from sites on each stream is quite similar up to the second summer of life (Fig. 49). As would be expected, it is at its greatest value during the initial life stages following emergence, from which it declines to lower levels in the late summer and fall. Results from the netting operations on Kelsey Creek indicate that few fish emigrated from the stream prior to December; thus the rate of loss during this period reflects only within-stream migration and

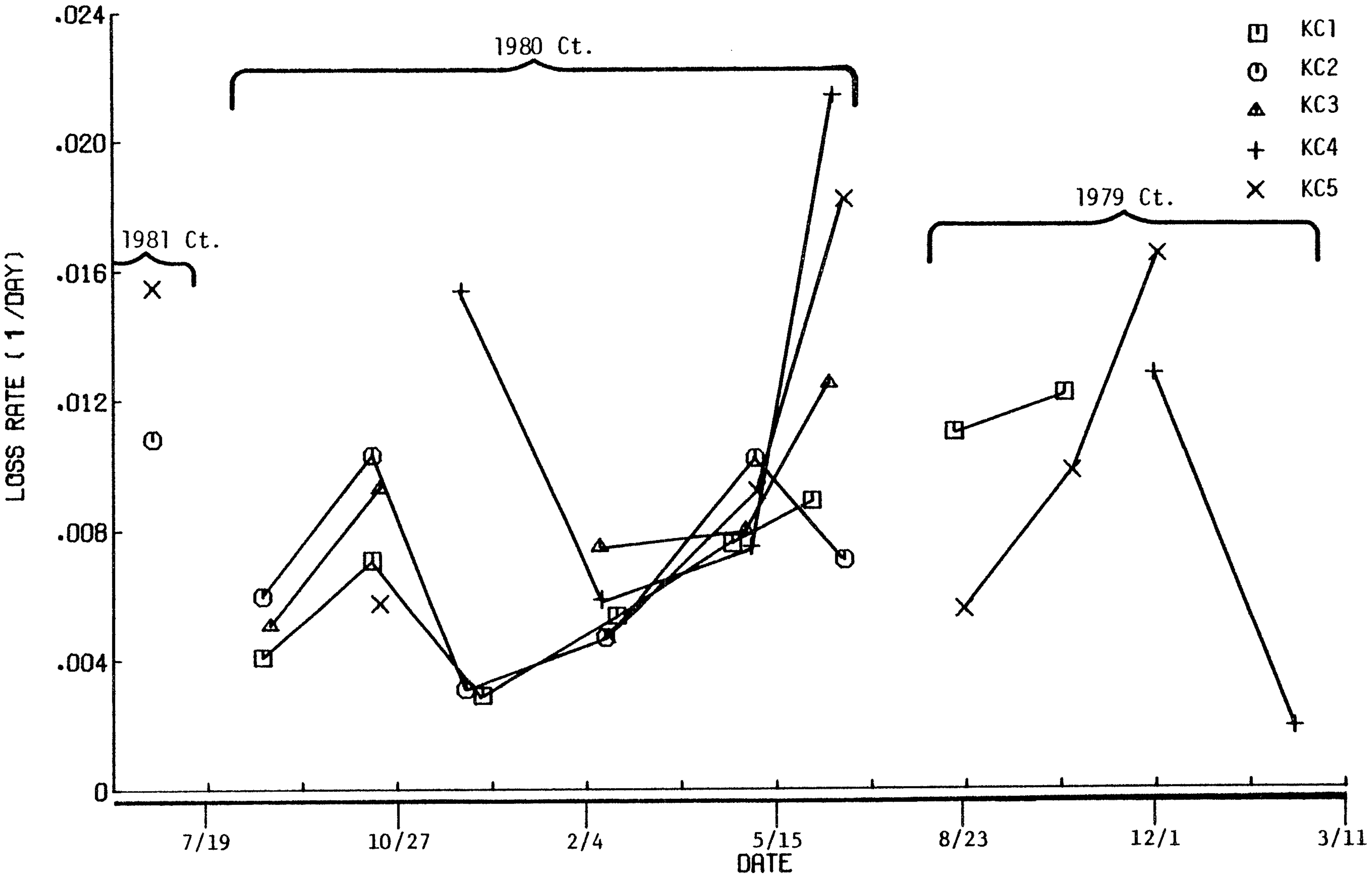


Figure 47. Loss rates for 1979 (right), 1980 (middle), and 1981 (left) year classes of cutthroat trout at sampling sites in Kelsey Creek.

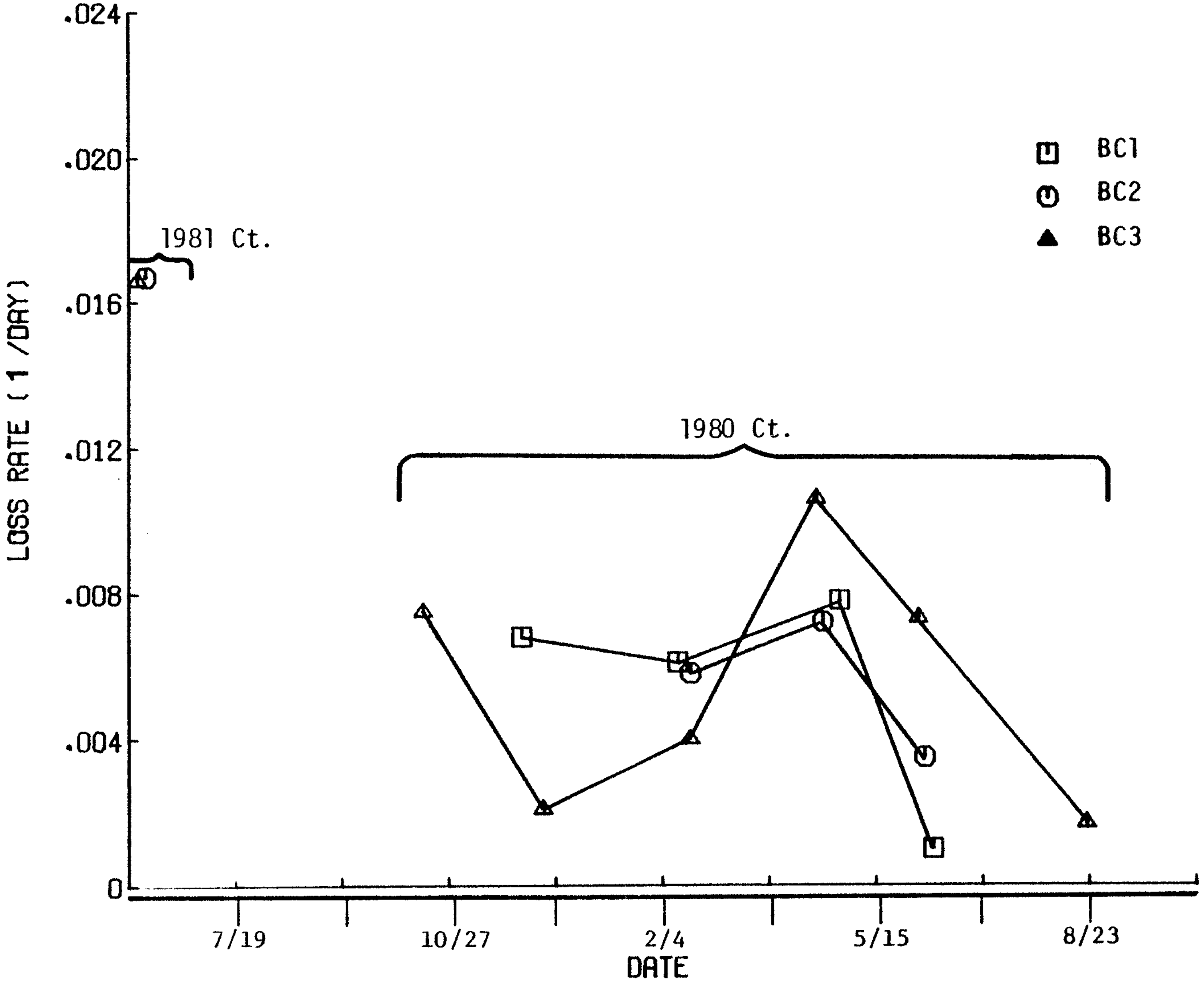


Figure 48. Loss rates for the 1980 (right) and 1981 (left) year classes of cutthroat trout at sampling sites in Bear Creek.

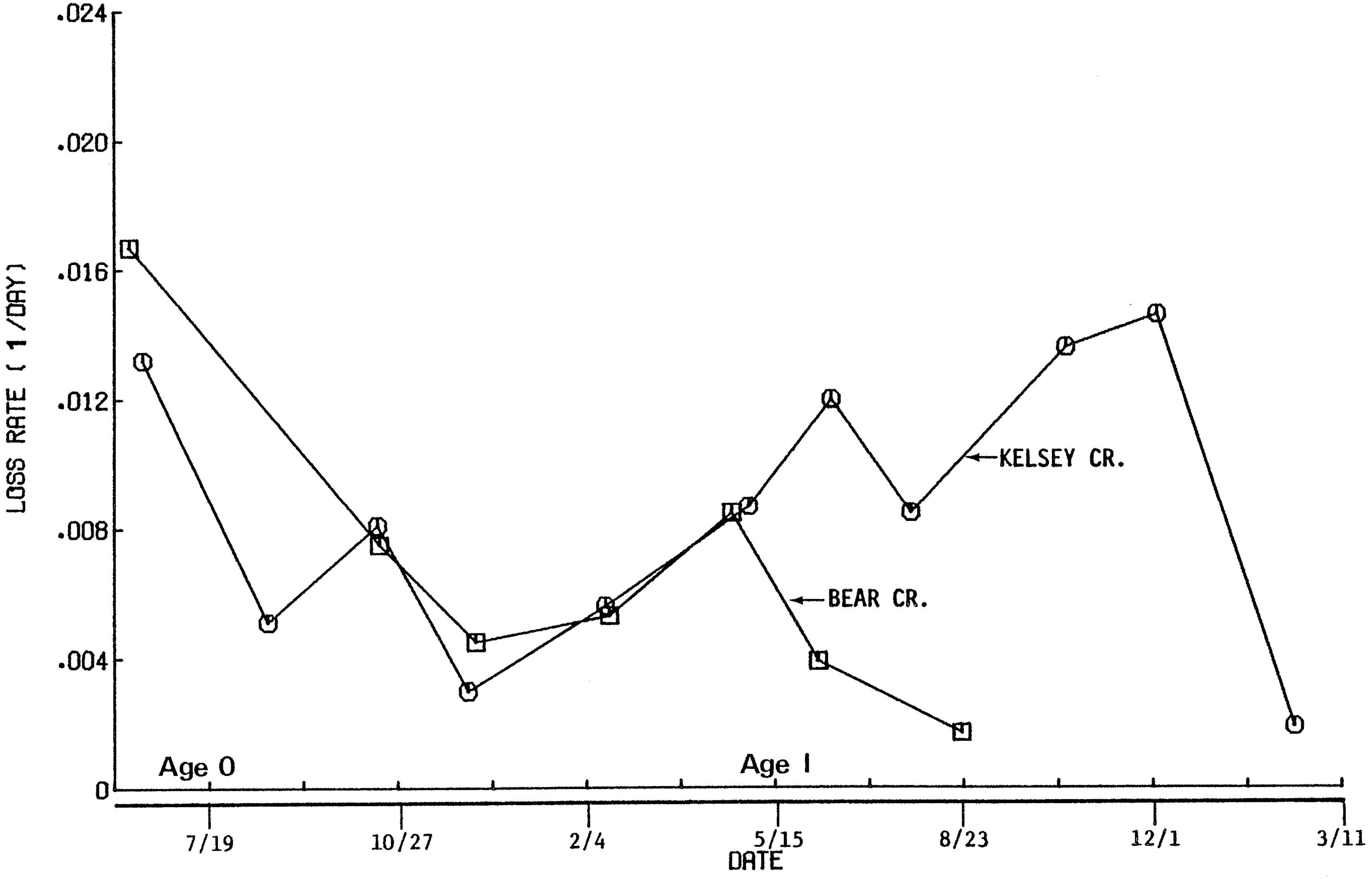


Figure 49. Mean loss rates for cutthroat trout at sample sites in Bear and Kelsey Creeks. Includes data from 1979, 1980, and 1981 year classes.

true mortality.

The slight rise in L during the period December to March, as well as the precipitous rise from March to May, undoubtedly results from the emigration of fish from the stream as well as mortality. The rate of loss of cutthroat trout in their second summer continued to be high in Kelsey Creek, while in Bear Creek, L dropped to .002/day, the lowest level observed during this study.

The rate at which marked coho salmon were lost from the study sites on Bear Creek is shown in Figure 50. Results from the 1980 cohort were variable between sites — the range of L observed exceeded .006/day for each sample period. The most rapid loss rate of coho salmon (.016/day) occurred with the 1981 cohort at BC1 in the period of March to May.

When the rate of loss is averaged over sites, the trend displayed in L for marked coho fingerlings in their initial 5 months of life may be seen to be similar to that previously described for cutthroat trout. However, in the period succeeding September, L reached a level in Bear Creek nearly equal to that observed for fry.

The abundance of the 1980 cohort of coho salmon in Kelsey Creek was not sufficient to allow the estimation of L. The mean rate of disappearance for the 1981 year class was similar to that observed in Bear Creek (Fig. 51).

7.5.3.2.2 Rate of Replacement

The replacement rate of marked cutthroat trout at the study sites on Kelsey and Bear Creeks is shown in Figures 52 and 53, respectively. Once more, it should be noted that data from several cohorts have been plotted on a single graph to simulate the life of a single cohort. Sites on Kelsey Creek, with the exception of KC4, replicated one another quite closely. Somewhat greater

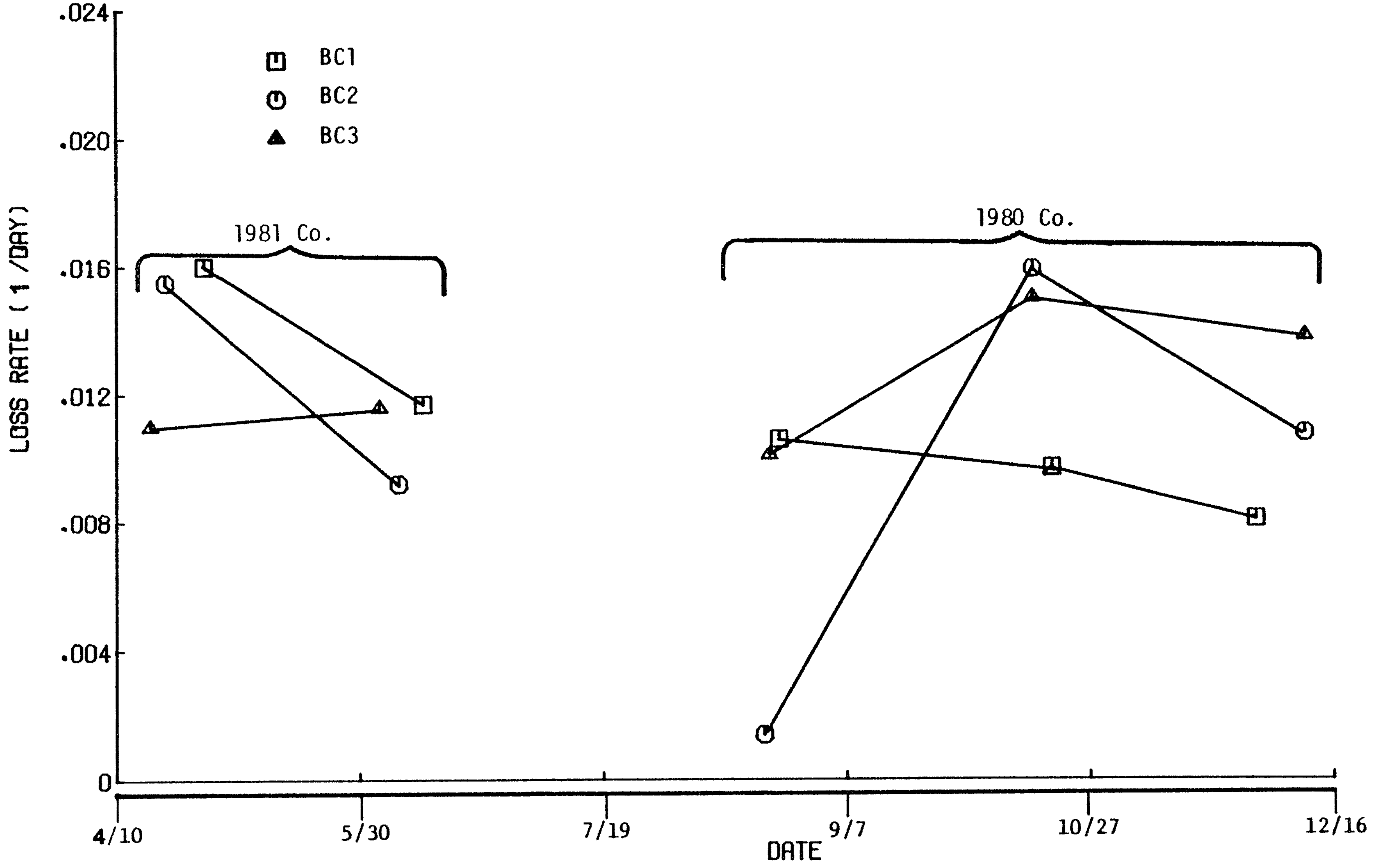


Figure 50. Loss rates for the 1980 (right) and 1981 (left) year classes of coho salmon at sampling sites in Bear Creek.

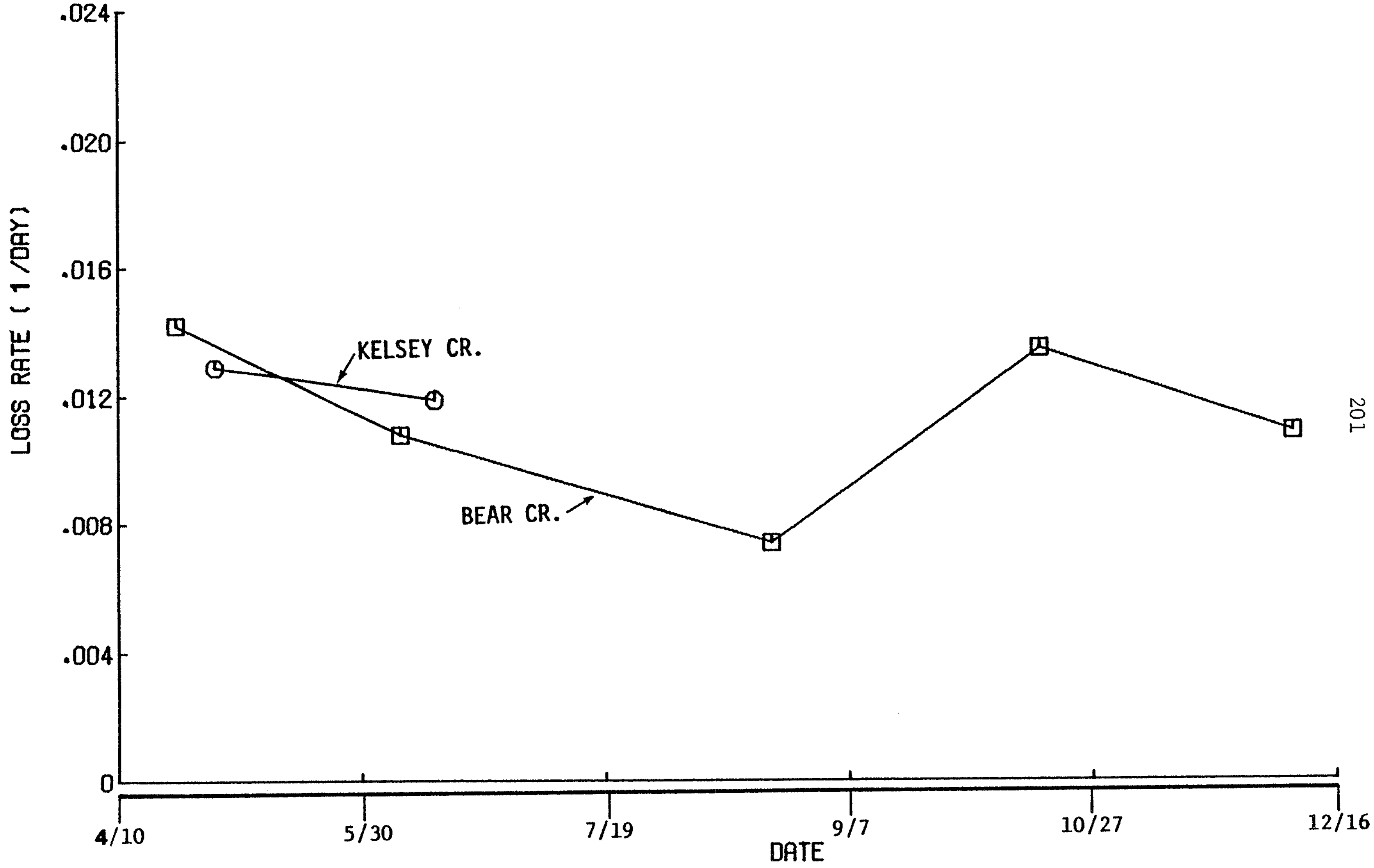


Figure 51. Mean loss rates for coho salmon at sample sites in Bear and Kelsey Creeks. Includes data from 1980 and 1981 year classes.

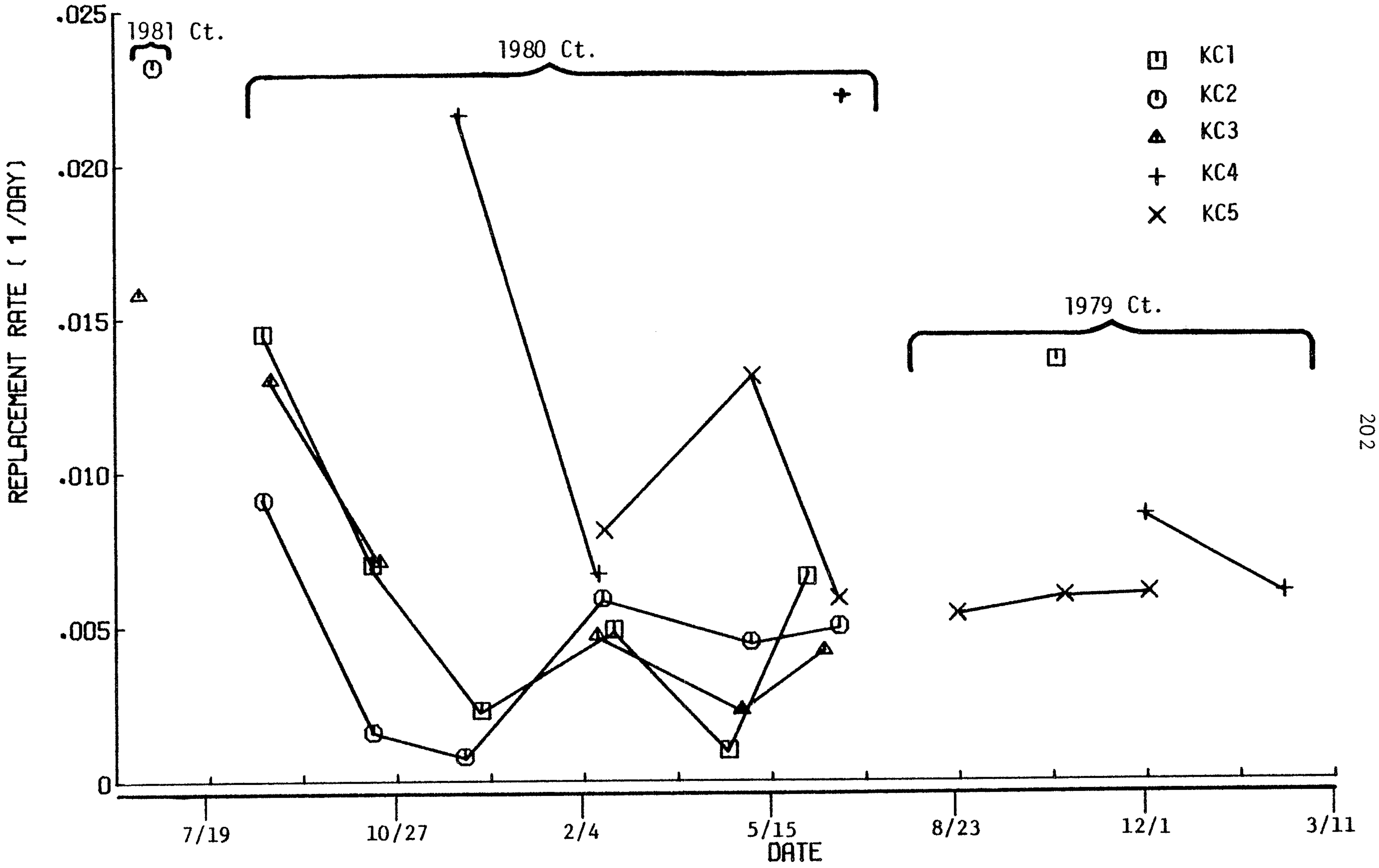


Figure 52. Replacement rates for the 1979 (right), 1980 (middle), and 1981 (left) year classes of cutthroat trout at sampling sites in Kelsey Creek.

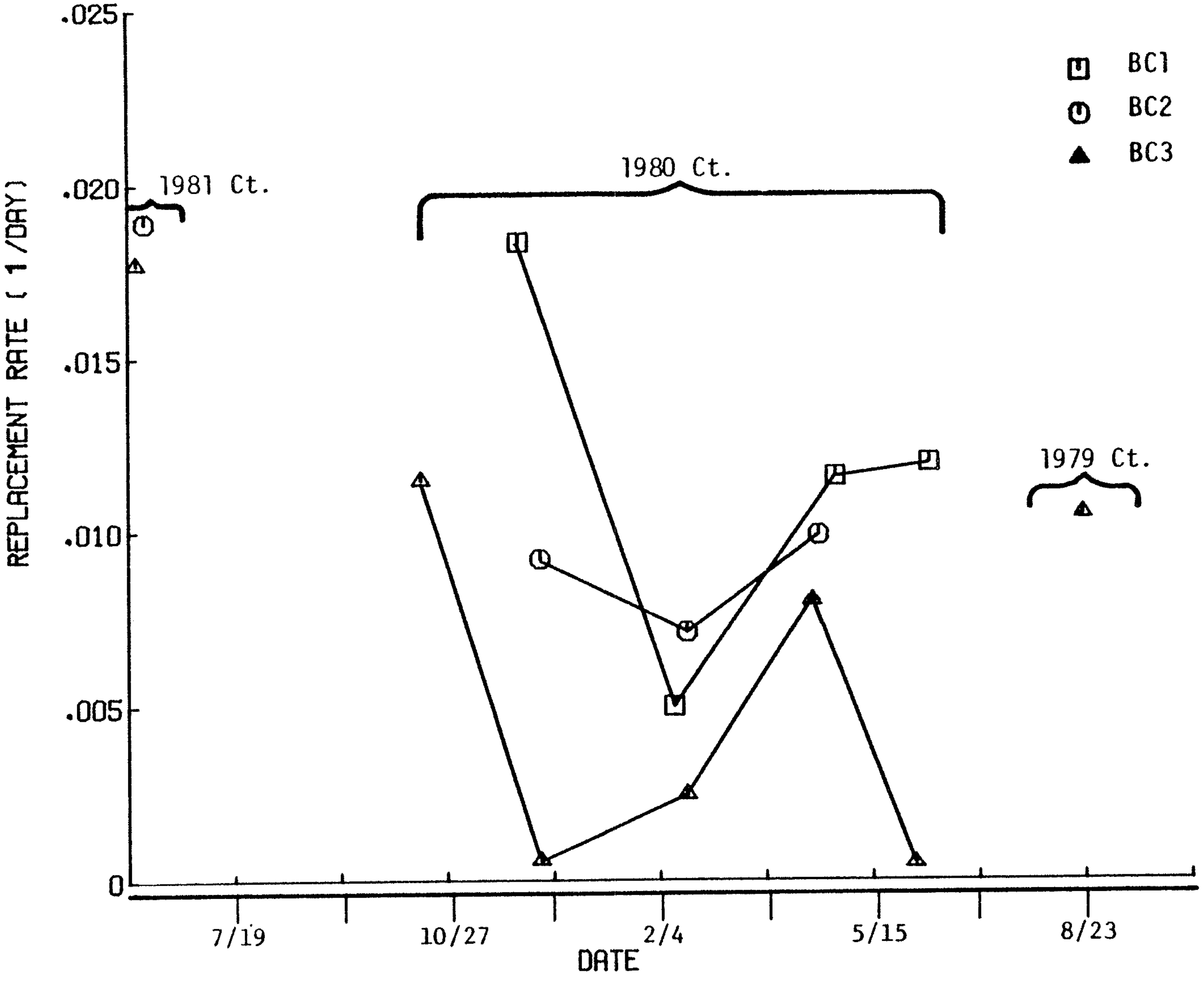


Figure 53. Replacement rates for the 1979 (right), 1980 (middle), and 1981 (left) year classes of cutthroat trout at sampling sites in Bear Creek.

variance existed in the R calculated for cutthroat trout in Bear Creek, which may be a reflection of the limited sample sizes.

The average replacement rate of cutthroat trout for each stream is plotted in Figure 54. In general, R is the greatest soon after emergence and declines throughout the remainder of the life of the fish. Seasonal trends are apparent in the increase of R during the spring months for age 0 fish and fall months for age I fish. Replacement rates for Kelsey Creek typically were below those for Bear Creek, particularly during the fall and winter of the first year of life.

The values of R calculated for the 1980 year class of coho in Bear Creek (Fig. 55) varied considerably among sites, but those for the 1981 cohort were relatively consistent. The mean replacement rates for coho salmon in Bear Creek (Fig. 56) declined only slightly during the summer and fall after emergence. The greatest decline was evident in the period of November to December, when the mean rate of replacement dropped by nearly 50 percent. The mean replacement rate for coho salmon in Kelsey Creek was consistently less than that observed in Bear Creek in the period from emergence to mid-summer. Thereafter sample sizes in Kelsey Creek were too small to allow the calculation of R.

7.5.3.3 Population Density — Variation within Streams

The number of fish inhabiting each study site after April 1980 was estimated by the removal method and, where possible, the Jolly-Seber mark-recapture model.

Population estimates calculated by each method, variances, and resultant pooled estimates are tabulated in Appendix Tables 15-30. Densities of the major species-age groups, expressed in terms of number of fish per m^2 of

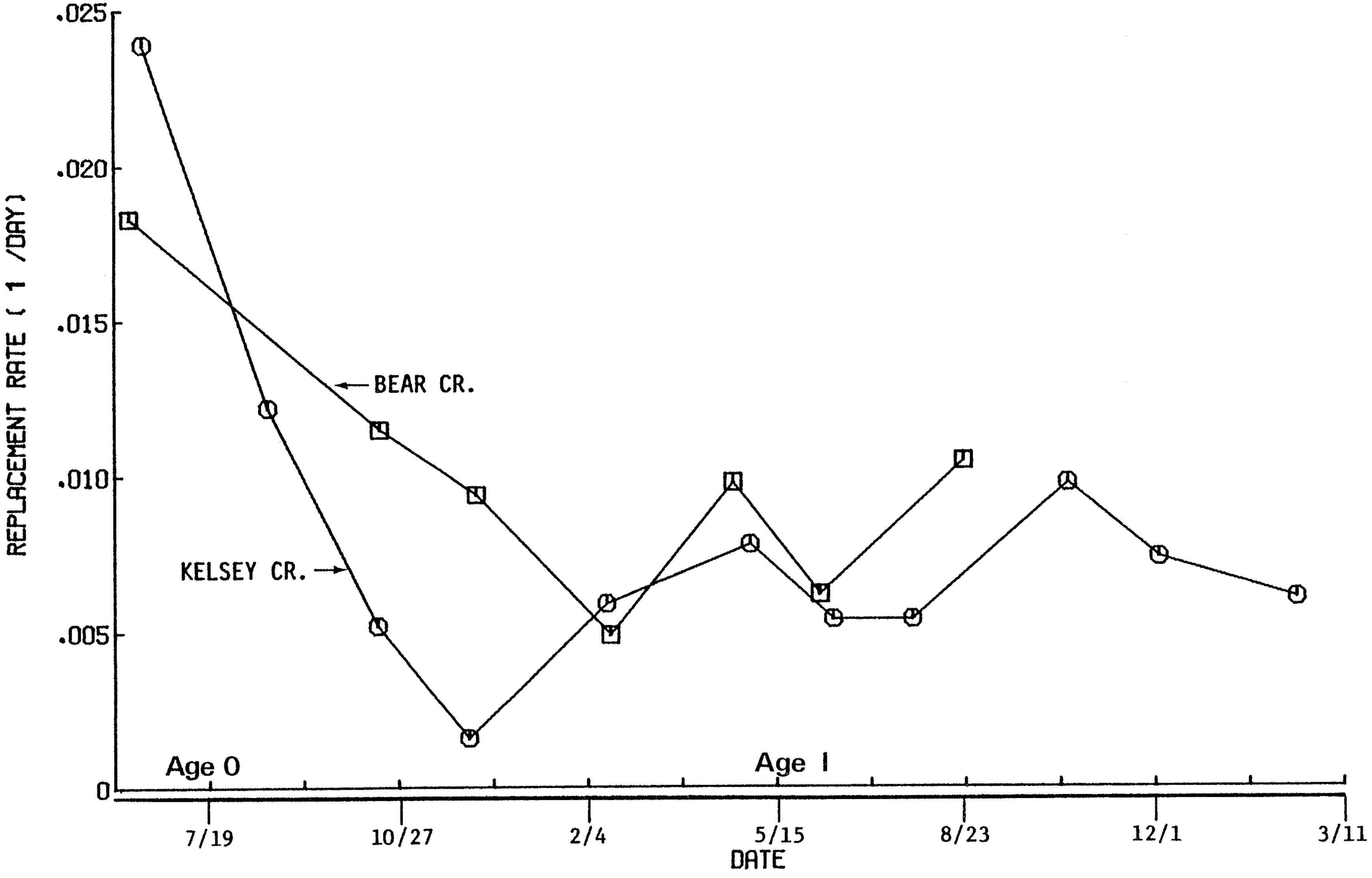


Figure 54. Mean replacement rates for cutthroat trout at sampling sites in Bear and Kelsey Creeks. Includes data from 1979, 1980, and 1981 year classes.

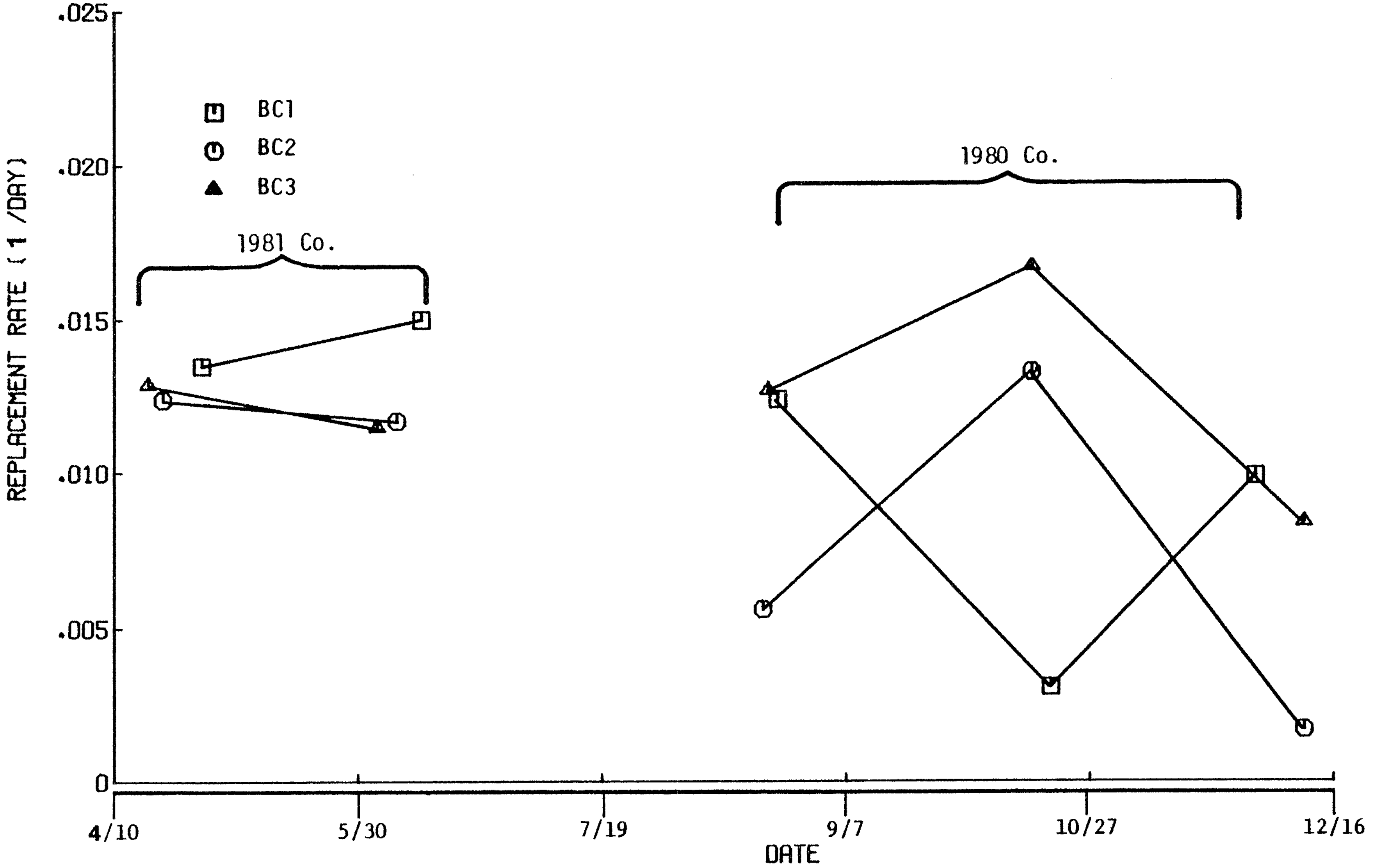


Figure 55. Replacement rates for the 1980 (right) and 1981 (left) year classes of coho salmon at sampling sites in Bear Creek.

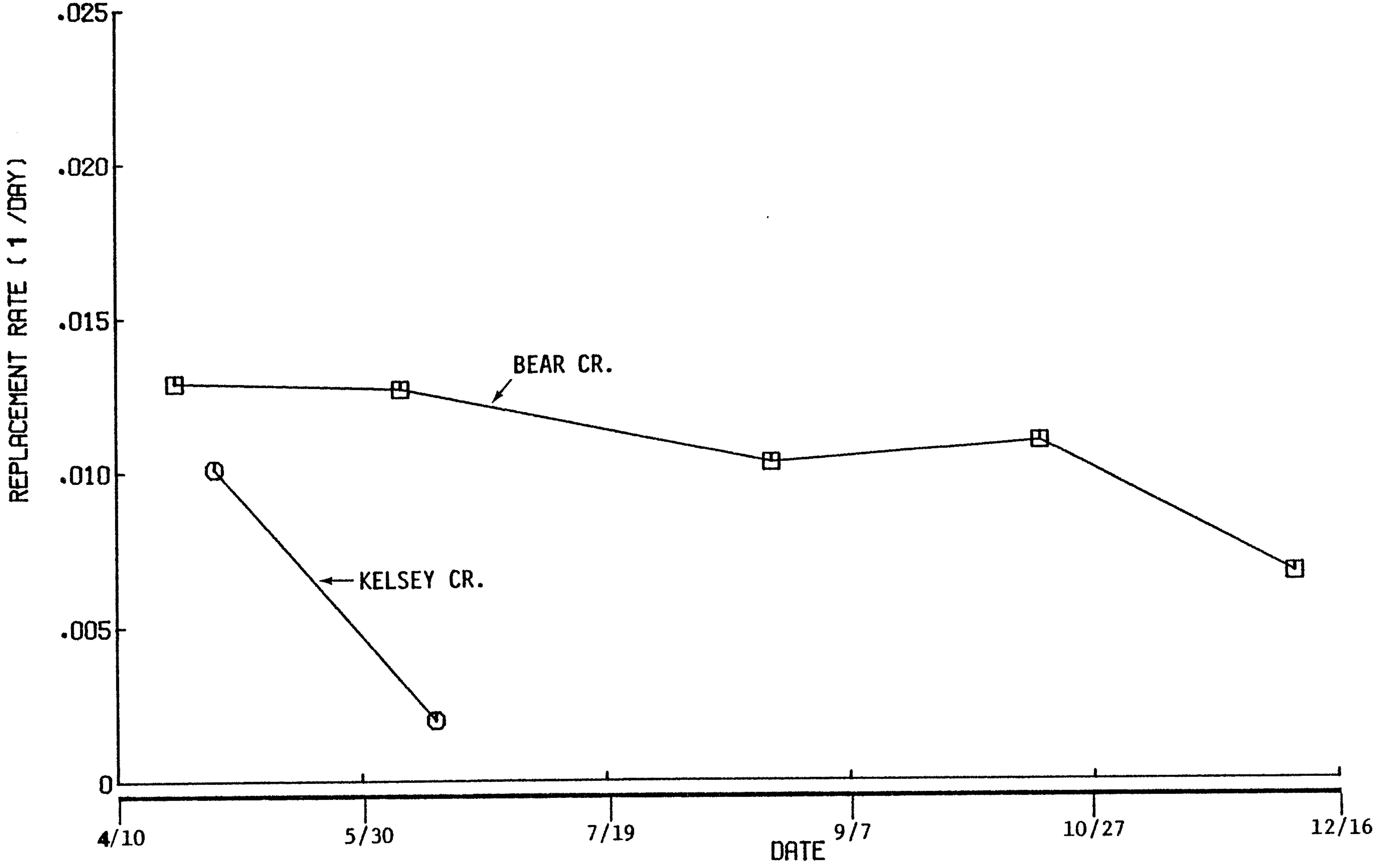


Figure 56. Mean replacement rates for coho salmon at sampling sites in Bear Creek. Includes data from 1980 and 1981 year classes.

low-flow stream surface area, are presented below. For each site, the density of fish at any given time may be seen to be an aggregate variable reflecting the basic processes of birth, death, immigration, and emigration which occurred prior to that time.

The density of the 1979 cohort of cutthroat trout in Kelsey Creek dropped rapidly at all sites after the initial sampling in mid-April, 1981 (Fig. 57). Initial densities ranged from a minimum of .13 fish/m^2 at KC3 to a maximum of .36 fish/m^2 at KC2. By the following sampling date in mid-July, the maximum density was observed as .14 fish/m^2 (KC1) while at KC4 no age I (1979 year class) cutthroat trout were captured. Densities were consistently low on all succeeding sampling dates. As would be predicted from the near absence of age III fish in the outmigrant net, only a very limited number of the 1979 year class were still present at the study sites in the summer of 1981.

The emergence of the 1980 year class of cutthroat trout in Kelsey Creek had scarcely begun when sampling commenced in mid-April 1980 (Fig. 57). By July of that year, the expected influx of new recruits was evident at KC2, where the population density of age 0 fish was estimated at .89 fish/m^2. Initial stocking densities at other sites were quite low in comparison to KC2 and to the initial size of the 1981 year class in the following year as well (reported below). The number of age 0 cutthroat trout captured at KC4 and KC5 remained at very low levels until December 1980, when densities were still less than 6 fish/m^2.

The estimates for the rate of loss (L_i) and replacement (R_i) previously presented may be used to determine the processes which underlie any alterations in population density. One such trend of interest is the rapid drop in the density of the 1980 cohort at KC2 after July 1980. Reference to Figures 47 and

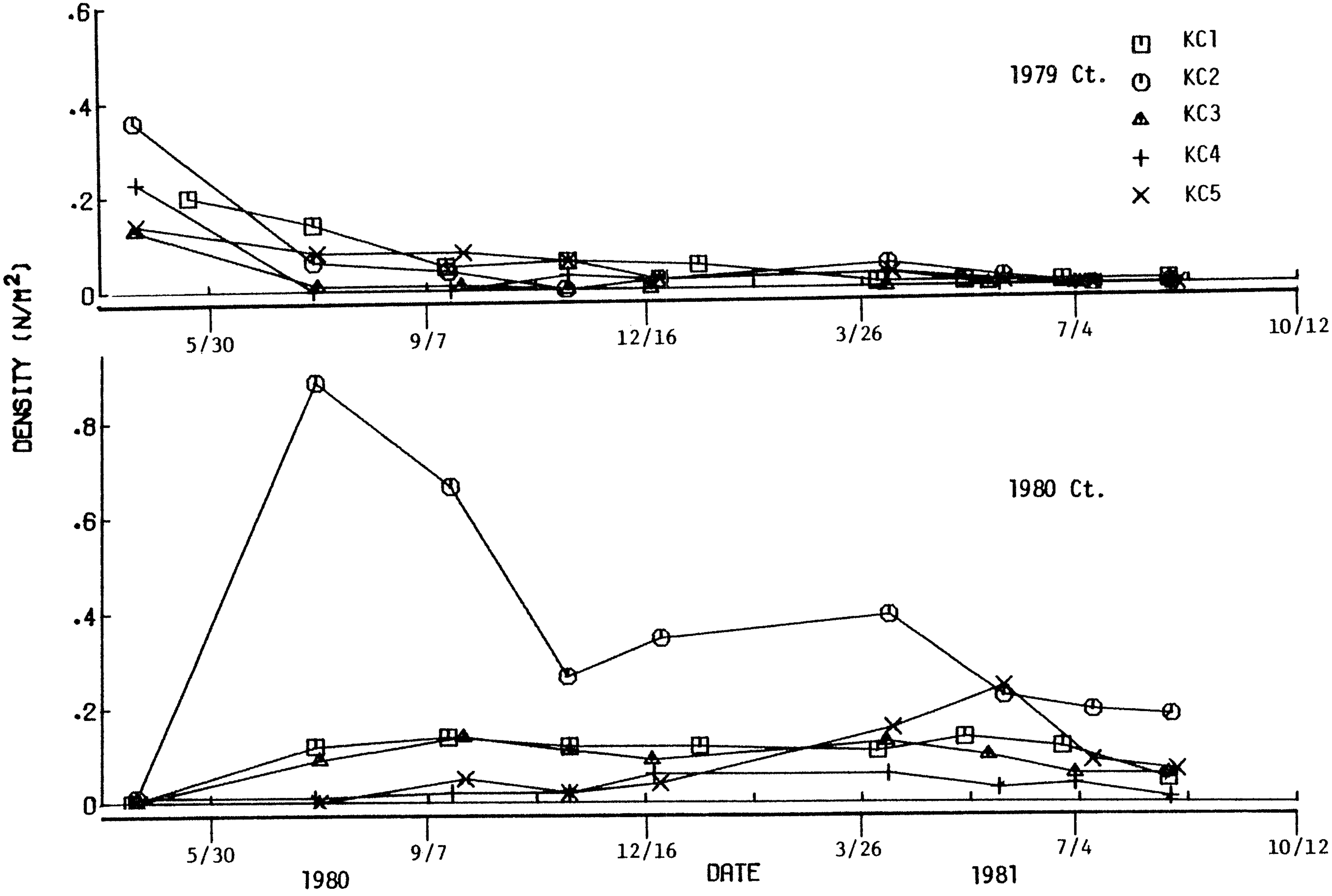


Figure 57. Density of the 1979 (upper) and 1980 (lower) year classes of cutthroat trout at sampling sites in Kelsey Creek, April 1980 to August 1981.

52 indicates that this is due to a low replacement rate as well as a loss rate high in comparison to that exhibited at the other sites. Also of interest is the increasing abundance of the 1980 cohort at KC5 through the winter and spring of 1981, which may be attributed to a high replacement rate coupled with an average loss rate.

Initial stocking densities of the 1981 cutthroat trout cohort were very high at KC2, KC3, and KC4, reaching a maximum of 1.43 fish/m^2 at the latter of these sites (Fig. 58). A dichotomy between the population dynamics of KC2, KC3, KC4 and KC1, KC5 was evident. The lag in population growth exhibited by KC1 and KC5 suggests that recruitment to these areas is primarily by immigration, whereas the other sites may be stocked by fry emerging from redds near or within the boundaries of the study sites. These conclusions could not be validated by the mark-recapture methodology due to the termination of field sampling in August of that year.

Coho salmon were considerably less abundant than cutthroat trout in Kelsey Creek. Initial stocking densities of the 1980 cohort were extremely low at all sites, the maximum observed being only .0325 fish/m^2 at KC1. By July of that year, coho were present only at site KC1 and KC3. No coho salmon were captured during the November sampling period, and only very limited numbers thereafter.

Coho salmon were slightly more abundant in Kelsey Creek in 1981, reaching a maximum of .17 fish/m^2 at KC3 in April (Fig. 59). Other sites typically supported less than 50 percent of the density observed at KC3 on all but the August sampling date. At that time the density of coho salmon ranged from 0 fish/m^2 at KC1, KC2, and KC5, to .04 fish/m^2 at KC3.

Variations between sites on Kelsey Creek in the total density of salmonids

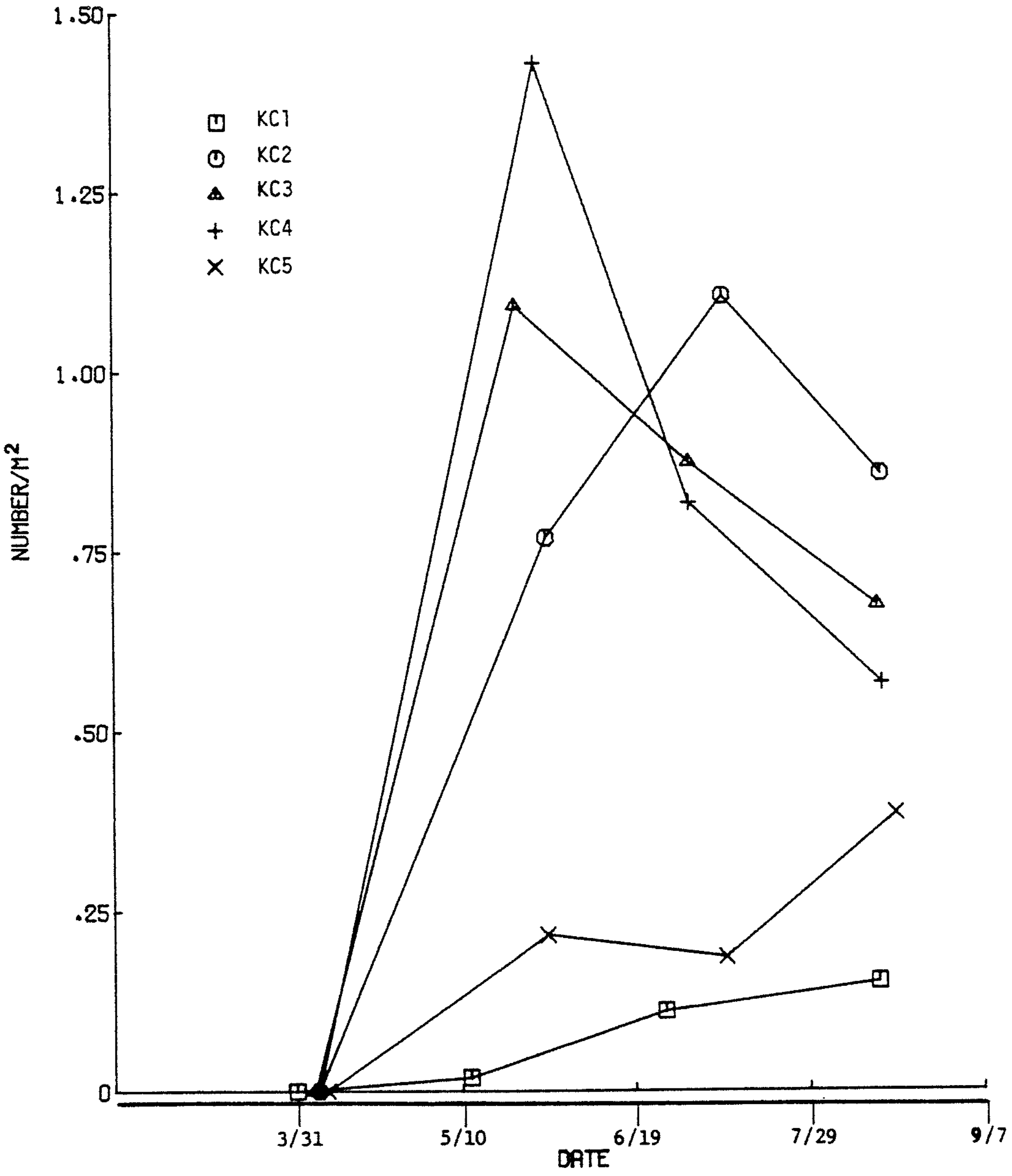


Figure 58. Density of the 1981 year class of cutthroat trout at sampling sites in Kelsey Creek.

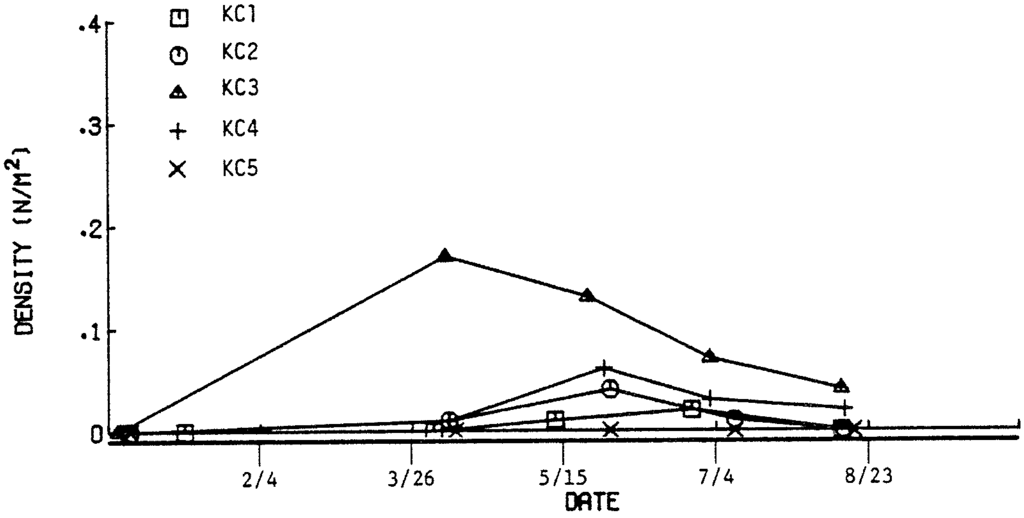


Figure 59. Density of the 1981 year class of coho salmon at sampling sites in Kelsey Creek.

exceeded an order of magnitude on several occasions (Fig. 60). Throughout the 1980 sampling season, the density of salmonids at KC2 was much greater than that observed at any other site, while densities at KC4 were generally considerably less. Only a single salmonid was captured at KC4 in July of 1980, and only 4 during the following sample session in September. The density of salmonids increased at all sites on Kelsey Creek in 1981, reflecting the strength of the 1981 year class of cutthroat trout.

The density of the 1979 cohort of cutthroat trout in Bear Creek varied between 0 and .08 fish/m^2, with BC3 consistently supporting the greater densities until April 1981 (Fig. 61). Densities of the 1979 year class after that date (age II fish) were low at all sites.

Members of the 1980 year class of cutthroat trout in Bear Creek were first captured in mid-July 1980 (Fig. 61). As was true for the 1979 cohort, densities were greatest at BC3, reaching a maximum of .32 fish/m^2 in November. It is interesting to note that the relative abundance of cutthroat at BC3 may be attributed to the excellent survival of fish residing there, rather than repeated restocking by immigrants, as revealed by the L and R statistics (Figs. 48 and 53). The density of the 1980 year class varied only slightly between the sites on Bear Creek by August of 1981, ranging from .05 fish/m^2 at BC2 to .11 fish/m^2 at BC3.

The initial stocking densities of cutthroat trout in Bear Creek were somewhat higher in 1981 at BC2 and BC3, reaching a maximum of .23 fish/m^2 at BC2 and .50 fish/m^2 at BC3 (Fig. 62). By midsummer, however, the densities were significantly lower, and approximated those found in the previous year.

The density of age 0 coho salmon estimated to inhabit Bear Creek in April of 1980 ranged from .26 fish/m^2 at BC3 to .61 fish/m^2 at BC2 (Fig. 63).

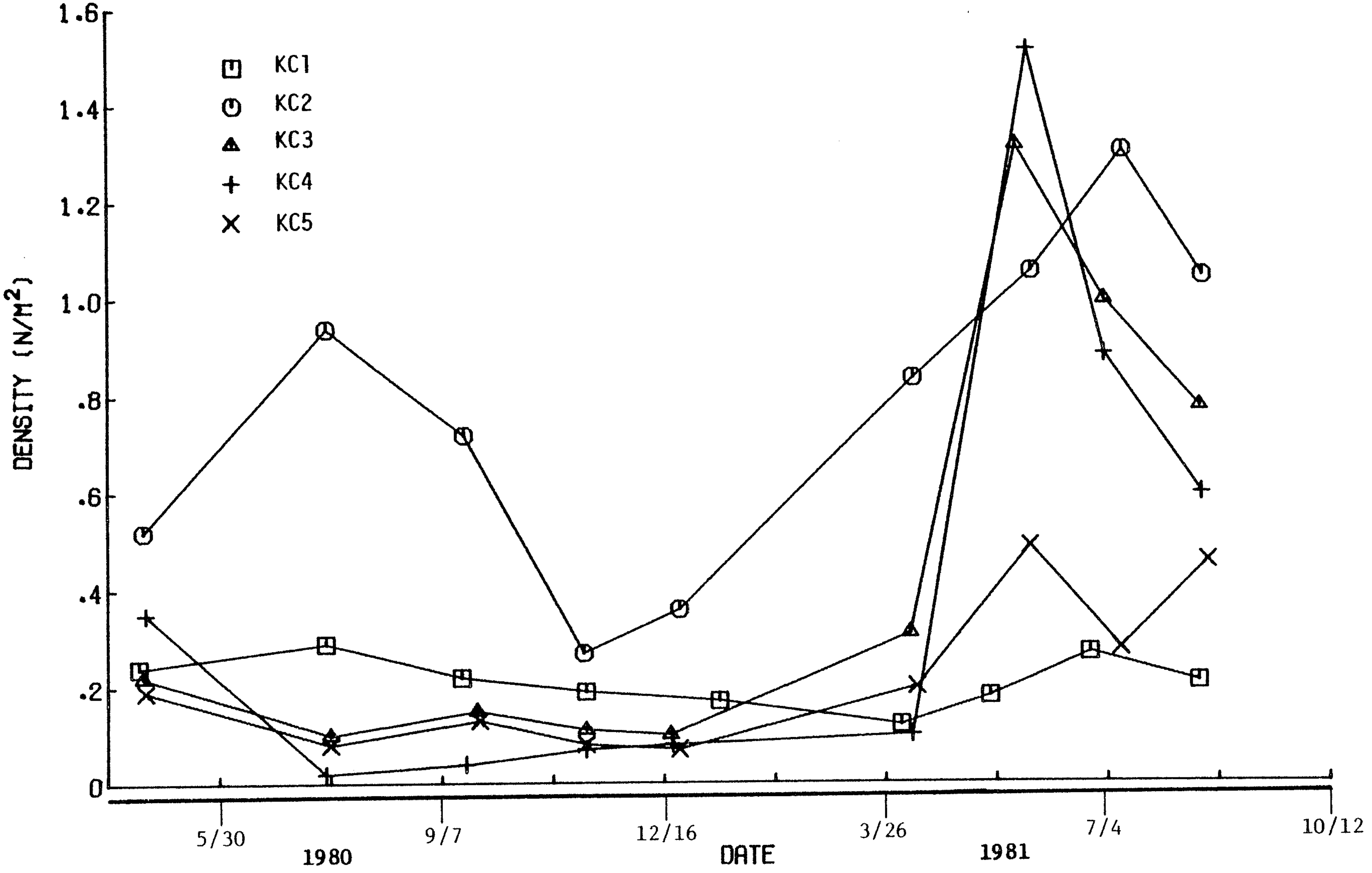


Figure 60. Total salmonid density at sampling sites in Kelsey Creek, April 1980 to August 1981.

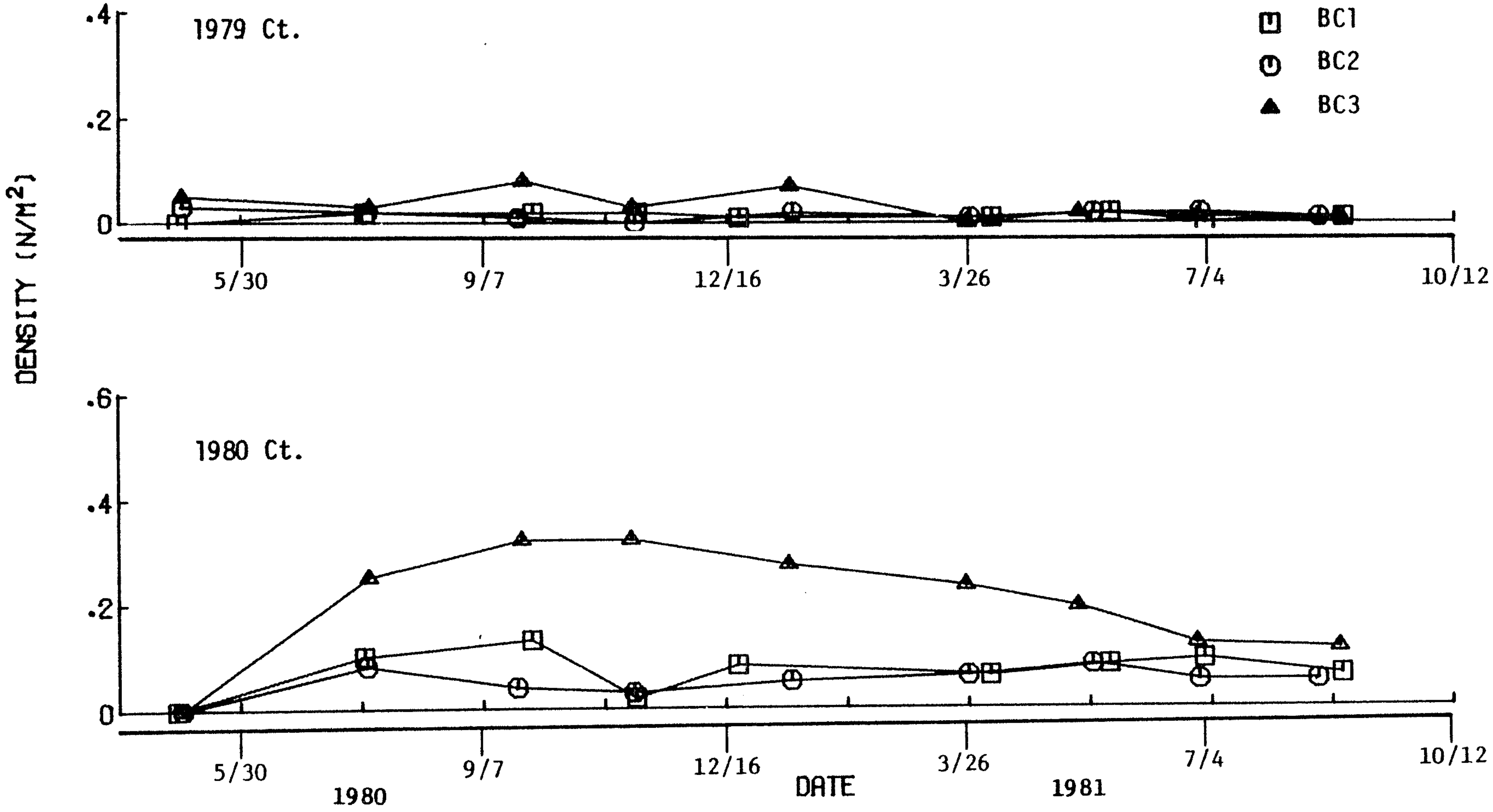


Figure 61. Density of the 1979 (upper) and 1980 (lower) year classes of cutthroat trout at sampling sites in Bear Creek, April 1980 to August 1981.

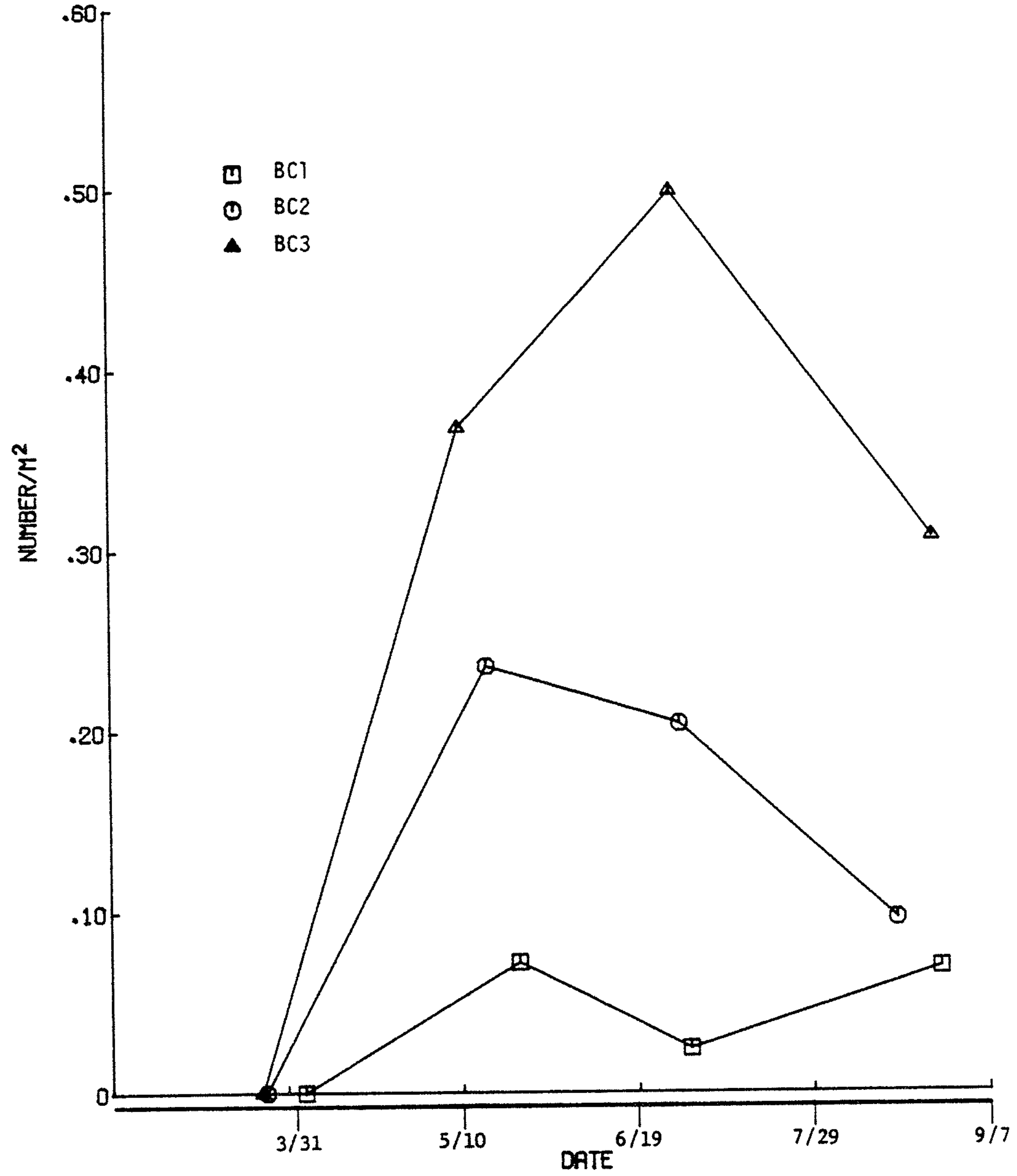


Figure 62. Density of the 1981 year class of cutthroat trout at sampling sites in Bear Creek.

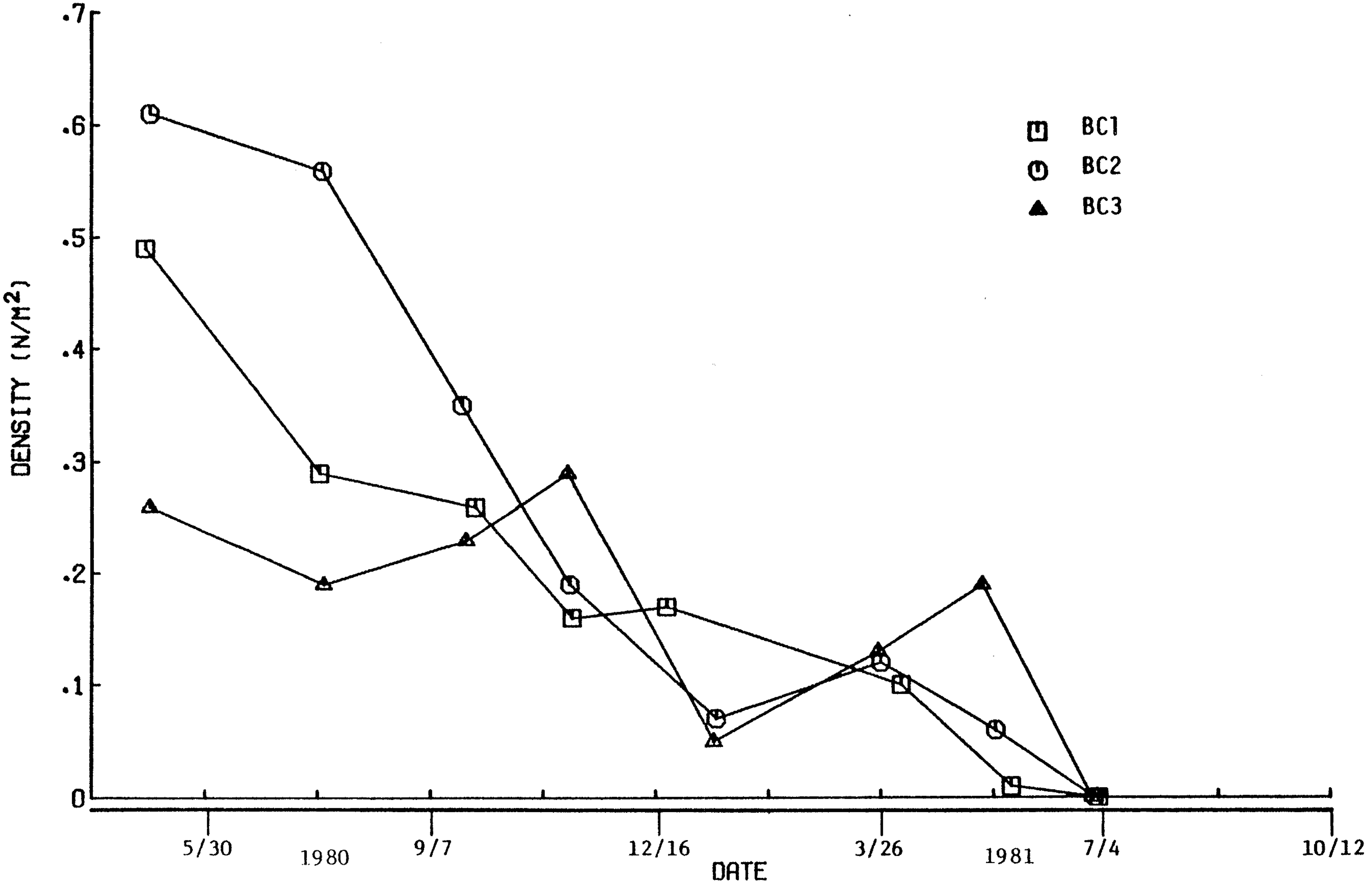


Figure 63. Density of the 1980 year class of coho salmon at sampling sites in Bear Creek.

After this time, the densities of coho salmon declined steadily throughout the summer and fall at BC1 and BC2, but increased slightly at BC3. This was apparently due to an influx of immigrants; the rate of loss of fish from BC3 was near the upper limit of values observed at the other sites during this period. As would be expected, when the rate of immigration into BC3 subsided in December of 1980, the density declined sharply. The 1980 year class of coho salmon was not captured in Bear Creek after May 23, 1981.

The 1981 cohort of coho salmon in Bear Creek was similar in many respects to that of 1980. As was true in the initial year of the study, the density of coho salmon was greatest at BC2 (1.02 fish/m^2) and least at BC3 (.29 fish/m^2) during the April sampling dates (Fig. 64). The basic population dynamics exhibited at BC2 and BC3 varied little from the previous year as well, with the number of fish estimated to reside in BC3 dropping rapidly throughout the spring and summer while the population density in BC3 remained stable. Note that in this case, however, the greater loss rate of BC2 is the factor of importance, as the replacement rates differed very little between these sites. By the first sampling date in August, the abundance of the 1981 cohort varied only from .33 fish/m^2 at BC3 to .48 fish/m^2 at BC1.

As was true in Kelsey Creek, the total salmonid density varied considerably between sites on Bear Creek (Fig. 65). In both 1980 and 1981 densities were greatest at BC2 during the early summer, but greater at BC3 in later months. This was due in part to the greater recruitment of age 0 cutthroat at BC3, which emerged later than the coho salmon predominate at BC2. Differences in the loss and replacement rates for coho at each site also caused an increase in the coho population of BC3 in the late summer months. The total salmonid density increased at all sites on Bear Creek in 1981.

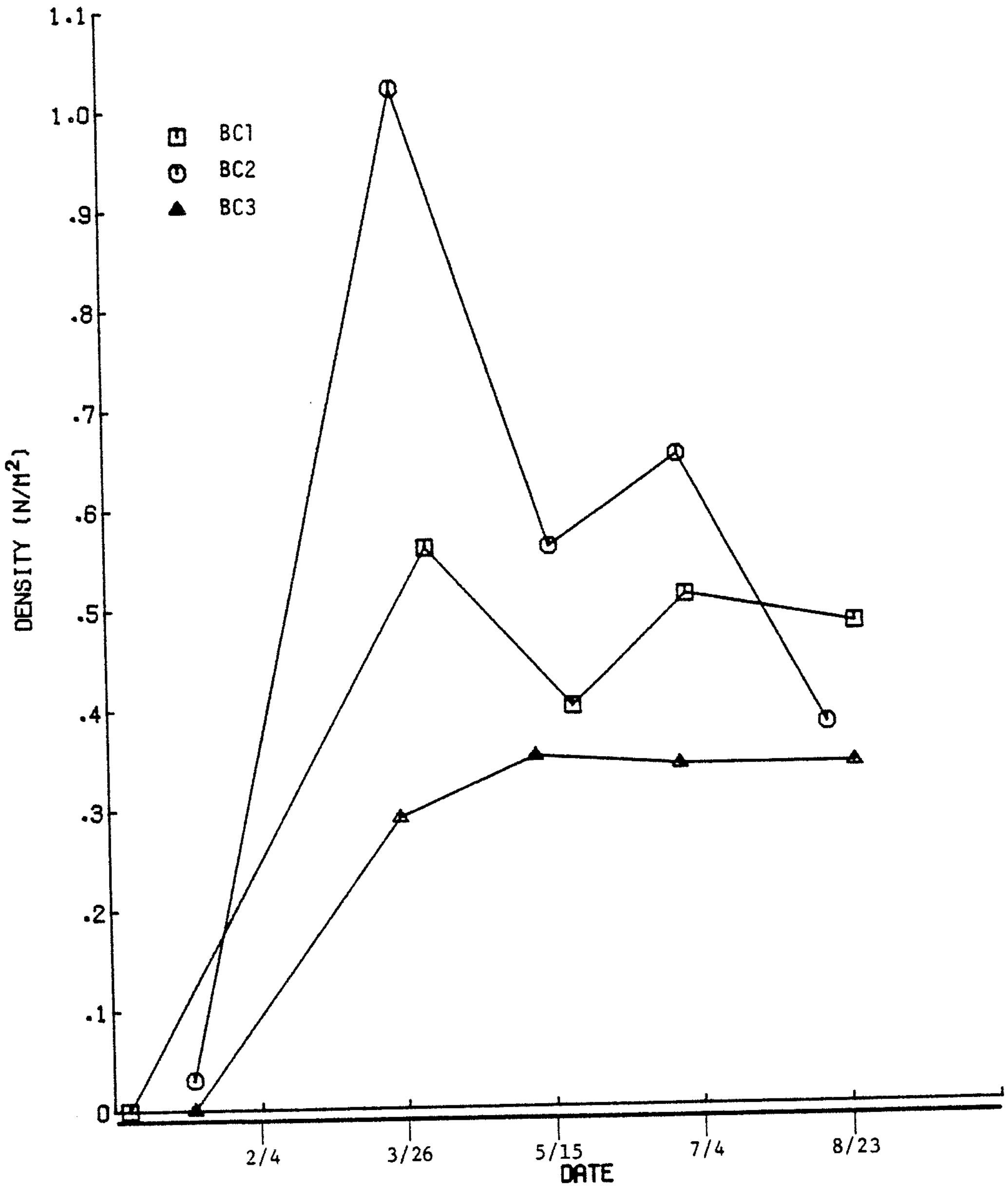


Figure 64. Density of the 1981 year class of coho salmon at sampling sites in Bear Creek.

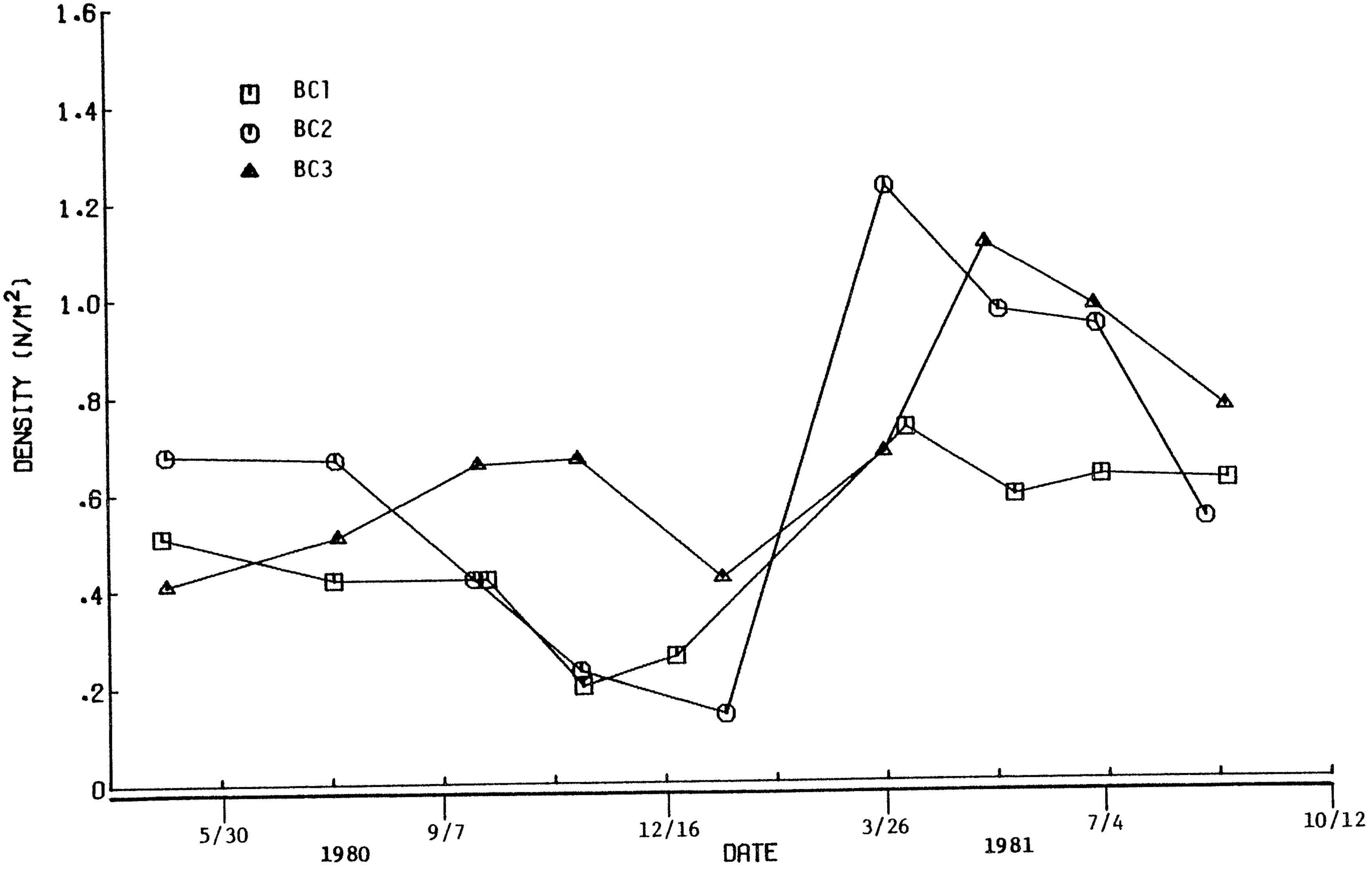


Figure 65. Total salmonid density at sampling sites in Bear Creek, April 1980 to August 1981.

7.5.3.4 Comparison of Salmonid Densities between Streams

As was previously alluded to, significant differences existed in the composition of the salmonid populations inhabiting each stream. Coho salmon comprised only a small fraction of the salmonids sampled in Kelsey Creek (Fig. 66), but they frequently exceeded 50 percent of the total salmonid population of Bear Creek (Fig. 67). Also of interest was the limited number of cutthroat trout older than age II inhabiting Kelsey Creek. Cutthroat of up to age III were found in Bear Creek, although in limited numbers.

Several trends are evident in a plot of the mean density of salmonids in each stream for the period April 1979 to August 1981 (Fig. 68). It is apparent that the abundance of salmonids in Kelsey Creek was reduced substantially in 1980 relative to both 1979 and 1981. The maximum density recorded in 1980, .29 fish/m^2, was less than 30 percent of that observed in 1979 and 1981. The salmonid population of Bear Creek during this three-year period was similarly unstable, as the density of salmonids increased in each succeeding year. Projecting these temporal trends of each stream upon one another, the density of salmonids in Kelsey Creek was greater than that in Bear Creek in 1979, less than in 1980, and very similar in 1981.

7.5.4 Biomass and Growth of Salmonids

Species length frequency histograms for each sampling date are presented in Appendix Figures 1-4. The logarithmic length and weight relationships for cutthroat trout and coho salmon sampled in 1979 are given in Table 33. Graphs of the length-weight relationship for each species in each stream are presented in Appendix Figures 5-8. Differences in the length-weight relationship were found in Kelsey Creek cutthroat above and below a length of 48 mm.

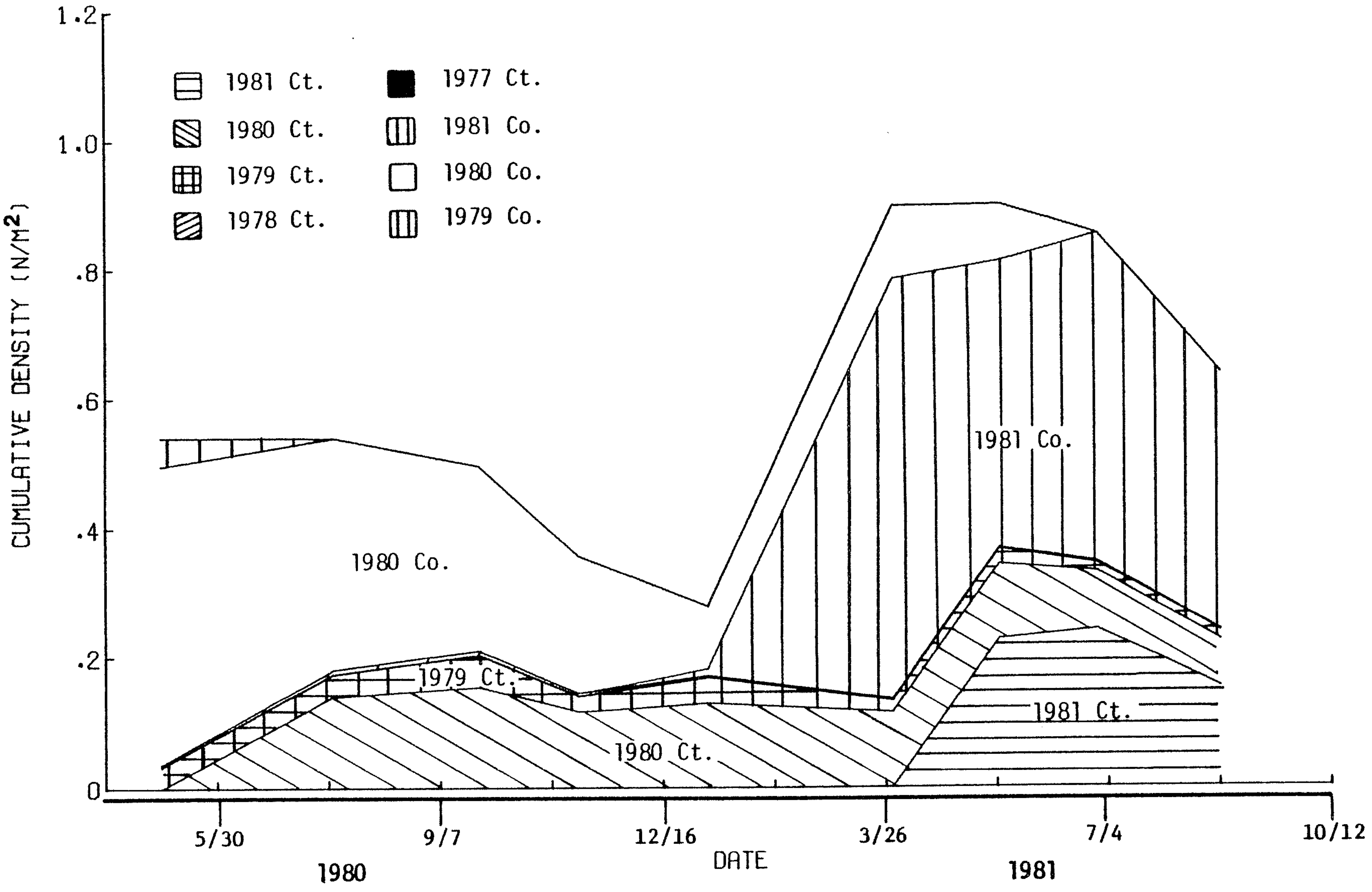


Figure 66. Mean density of salmonids, apportioned by species-age classes, at sampling sites in Bear Creek, April 1980 to August 1981.

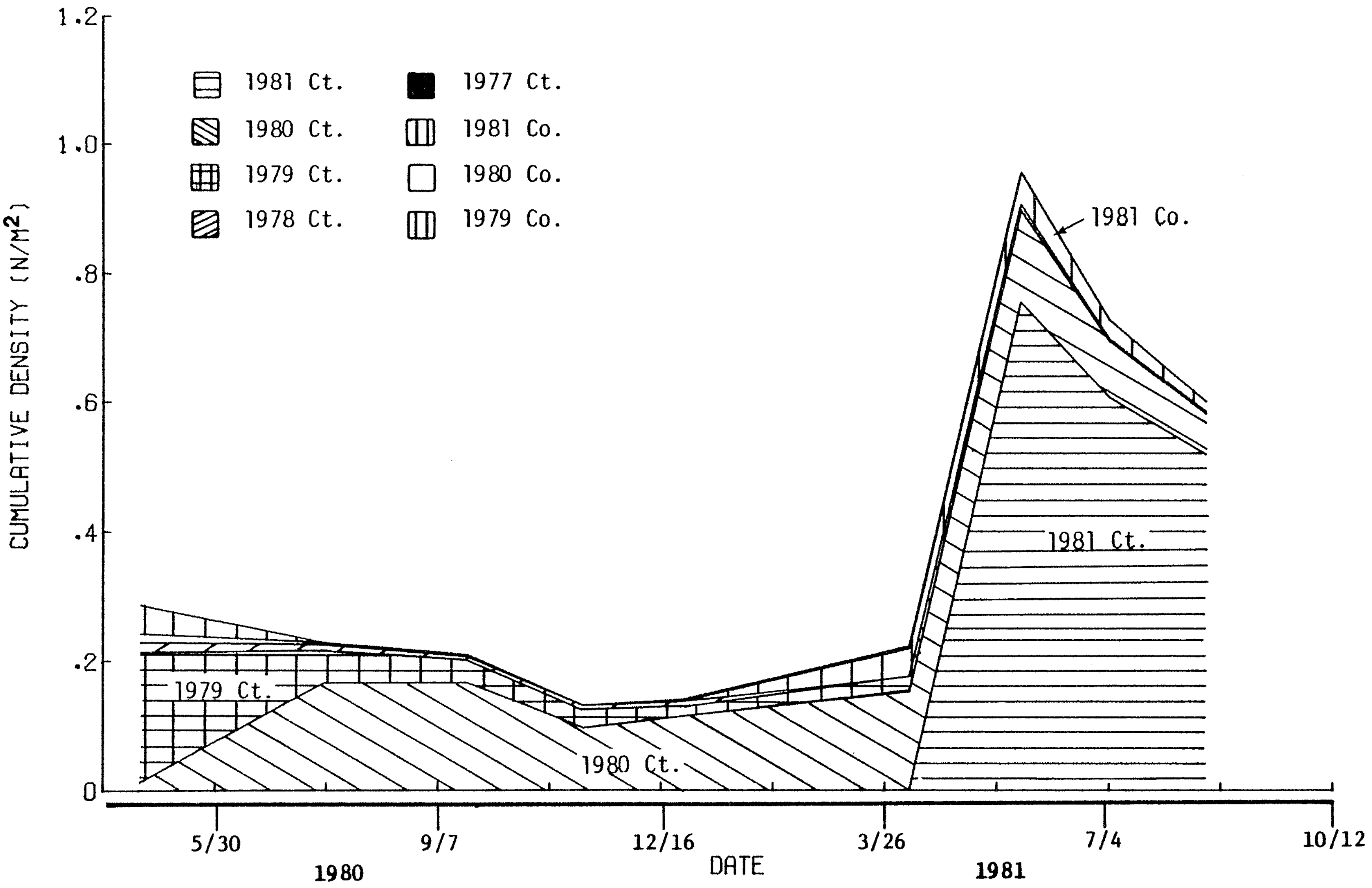


Figure 67. Mean density of salmonids, apportioned by species-age classes, at sampling sites in Kelsey Creek, April 1980 to August 1981.

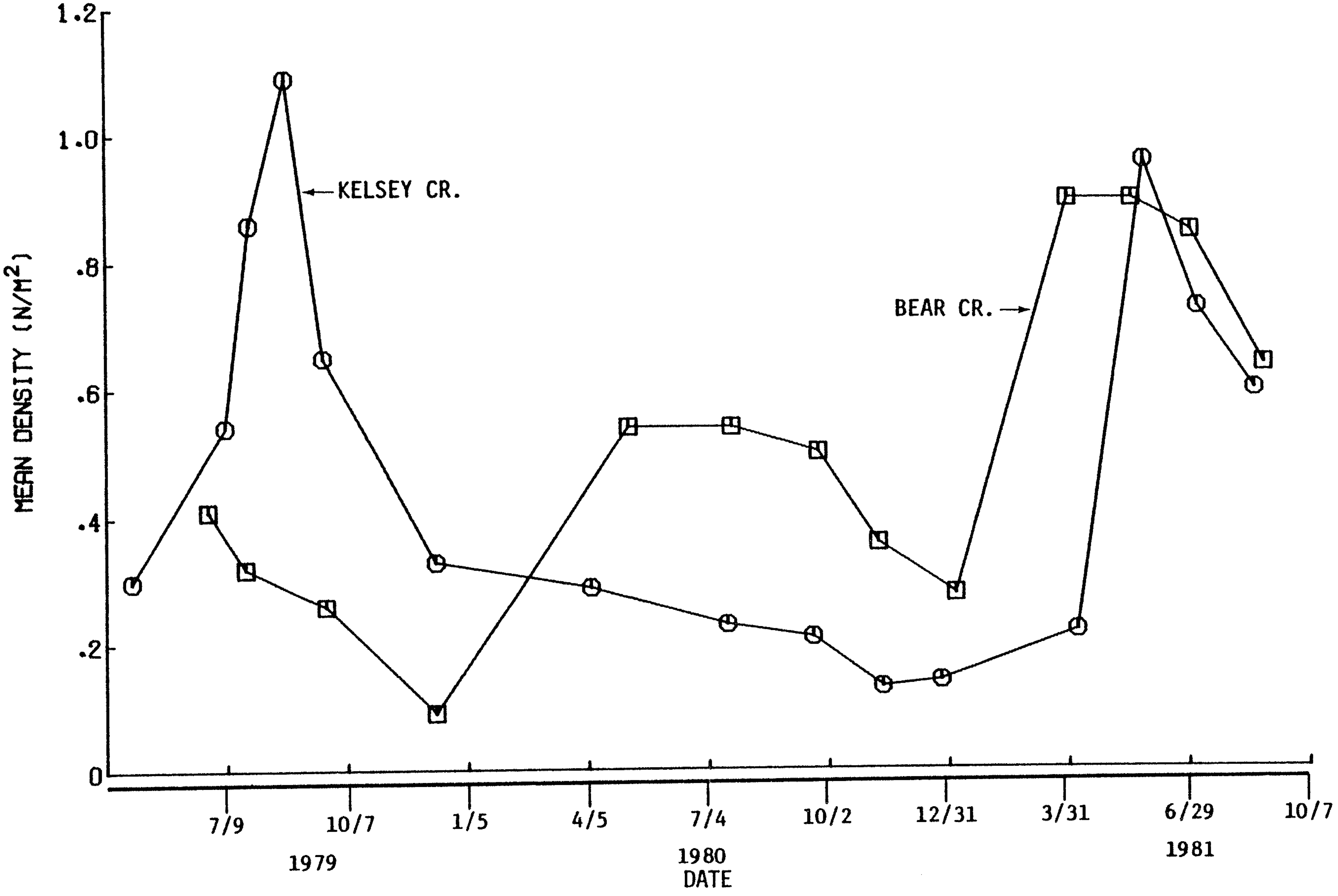


Figure 68. Mean density of salmonids in Bear and Kelsey Creeks, April 1979 to August 1981.

Table 33. Length-weight regressions for salmonids sampled in Bear and Kelsey Creeks, April 1979 to July 1980 (y = ln weight, x = ln length).

Stream	Species	Length	Sample size	Correlation coefficient		Equation						
Kelsey	Cutthroat	< 49	30	0.91	*	y = -6.469 + 3.926 x						
		> 48	1741	0.99		y = -5.356 + 3.167 x	0				0	
	Coho	All	207	0.97		y = -5.380 + 3.187 x						***
Bear	Cutthroat	< 55	28	0.94	***	y = -6.102 + 3.693 x						
		> 54	156	0.99		y = -4.840 + 2.909 x		*		***		
	Coho	< 55	31	0.86	**	y = -7.911 + 4.704 x			*			
		> 54	334	0.91		y = -4.495 + 2.729 x						

0 = No significant difference

* = 0.05

** = 0.01

*** = 0.001

Similarly, separate regression equations were determined for Bear Creek cutthroat and coho less and greater than 54 mm in length. Insufficient data prevented the separation of Kelsey Creek coho into "small" and "large" size groups. Weight changed more rapidly in relation to length in the small fish of both streams; the coefficient b for small fish was significantly higher than the cube in all cases.

Separate tests for equal intercepts and slopes were used to compare the regression lines resulting from the 1979-1980 data for 1) cutthroat and coho from the same stream, and 2) the same species from different streams. Only length-weight relationships for the same general size group (e.g., cutthroat above 48 and 54 mm) were tested for coincidence. As shown in Table 33, no significant difference was detected between large cutthroat and coho in Kelsey Creek, or between small cutthroat in both streams. Between stream comparisons of large fish of the same species reveal that larger Kelsey Creek cutthroat and coho (> 100 and 86 mm, respectively) are heavier than their Bear Creek counterparts.

In 1980-81, length-weight regressions were calculated independently for each stream, salmonid species, and sampling date. Within a stream, data were pooled for sampling dates whose regression lines did not differ significantly. The resultant regression equations, subsequently used to estimate fish weight from length, are listed in Table 34.

The estimated mean weight for each species-age group is tabulated by sample date in Appendix Tables 4 and 5. Time traces for the growth in length of the 1976-1981 cohorts of cutthroat trout in Bear and Kelsey Creeks for the period extending from May 1979 to August 1981 are plotted in Figures 69 and

Table 34. Length-weight regression equations utilized to estimate fish weight from length data measured in the field, April 1980 to August 1981.

Stream	Species	Date(s)	Data from	n	R^2	Regression equation
Bear Cr.	Coho	5/80, 7/80, 11/80 1/81, 3/81, 6/81 8/81	5/80, 7/80, 11/80 1/81, 3/81, 6/81	347	.93	$\ln(W) = -11.788 + 3.057 \ln(L)$
		9/80	5/80, 7/80, 9/80 11/80, 1/81, 3/81, 6/81	424	.91	$\ln(W) = -11.763 + 3.046 \ln(L)$
		5/81	5/80, 11/80, 1/81, 5/81, 8/81	276	.89	$\ln(W) = -10.091 + 2.666 \ln(L)$
Bear Cr.	Cutthroat	5/80, 7/80, 9/80 11/80, 3/81, 5/81, 6/81, 8/81	5/80, 7/80, 9/80 11/80, 3/81, 5/81 6/81, 8/81	488	.98	$\ln(W) = -11.098 + 2.908 \ln(L)$
		1/81	5/80, 9/80, 1/81	96	.97	$\ln(W) = -11.939 + 3.060 \ln(L)$
Kelsey Cr.	Coho	All dates	All dates	127	.97	$\ln(W) = -10.696 + 2.936 \ln(L)$
Kelsey Cr.	Cutthroat	4/80, 7/80, 9/80 12/80, 4/81, 7/81	4/80, 7/80, 9/80 12/80, 4/81, 7/81	557	.98	$\ln(W) = -11.648 + 3.026 \ln(L)$
		11/80	4/80, 9/80, 11/80 12/80	376	.97	$\ln(W) = -12.201 + 3.137 \ln(L)$
		5/81	5/81	89	.99	$\ln(W) = -12.679 + 3.277 \ln(L)$
		8/81	4/81, 7/81, 8/81	184	.98	$\ln(W) = -10.958 + 2.906 \ln(L)$

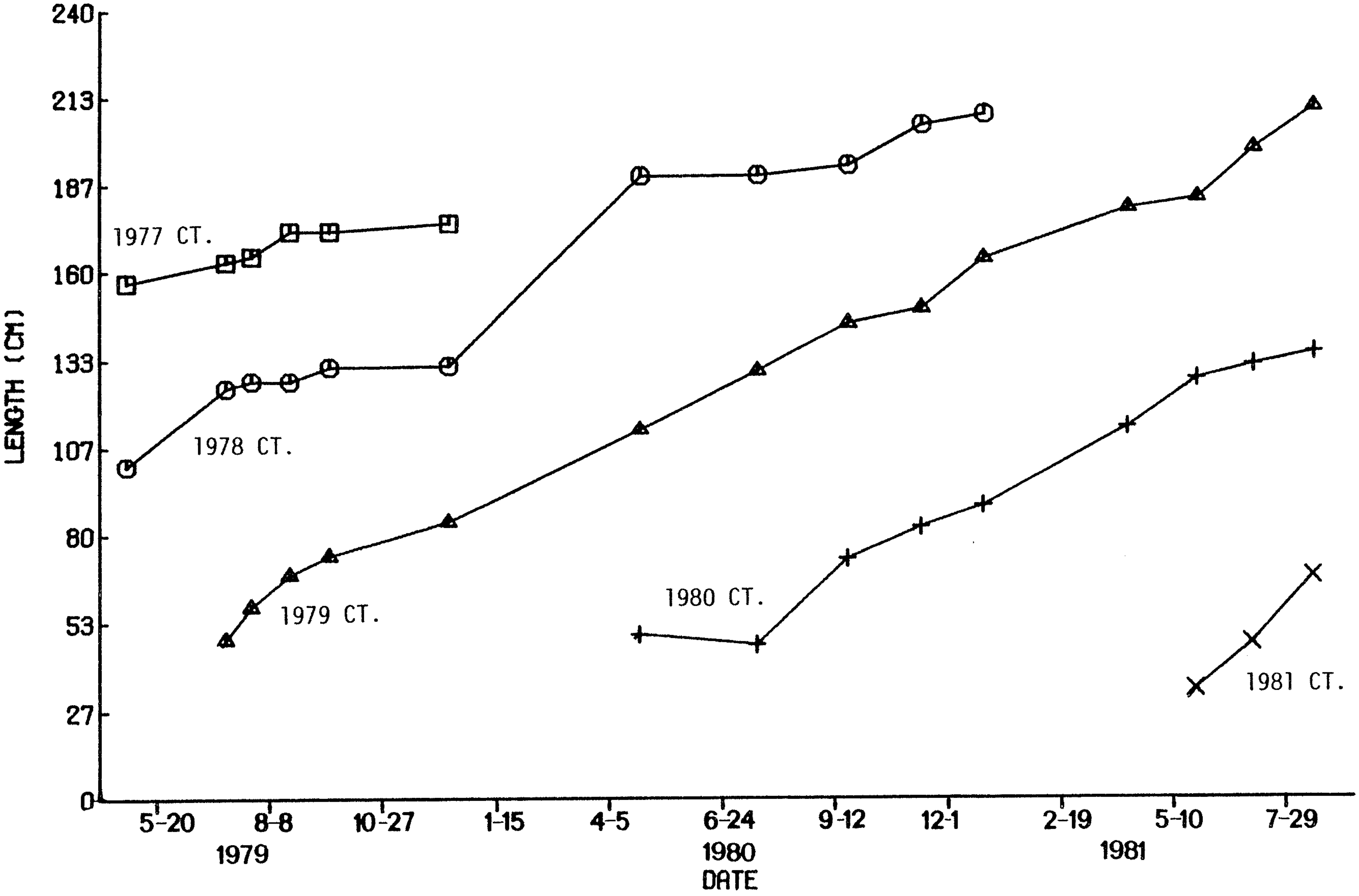


Figure 69. Mean lengths of the 1977 to 1981 year classes of cutthroat trout sampled in Kelsey Creek, April 1979 to August 1981.

70, respectively. Of interest in these figures are the steeper slopes of the time traces for Kelsey Creek, which indicates growth in length is more rapid than in Bear Creek. Graphs of the 1979, 1980, and 1981 cohorts of coho salmon (Fig. 71) display a similar result, with the mean length of Kelsey Creek coho salmon on any date exceeding those of Bear Creek by at least 10 cm.

Reasons for this disparity may be examined in greater detail by plotting the instantaneous rate of growth in weight for the population between sample dates. The daily instantaneous growth rates of cutthroat captured in 1979-1981 are plotted by age group at the midpoint of each sample interval in Figure 72 (see also Appendix Tables 4 and 5). Growth rates of age 0 cutthroat in Kelsey Creek in 1979, 1980, and 1981 were initially greater than Bear Creek fish of the same age, but growth in Kelsey Creek slowed considerably in the early fall. The rate of growth was also greater in Kelsey Creek during the spring months for age I fish of the 1979 and 1980 year classes. Similar observations appear applicable to newly emergent coho salmon in Kelsey Creek (Fig. 73), but firm conclusions after midsummer are precluded by the limited sample sizes available for Kelsey Creek.

A fundamental difference exists between the urban and control streams in the composition of the salmonid biomass supported by the available food resources. The biomass of salmonids, averaged over sites, supported by Kelsey Creek in 1980-81 is presented apportioned by species-age classes in Figure 74. Note that age 0 and I cutthroat trout account for the vast majority of biomass at all times, with coho salmon contributing a maximum of 10 percent. In contrast, the biomass of salmonids supported by Bear Creek in 1980-81 was distributed over a number of species-age classes (Fig. 75). Indeed, in September of 1980, the maximum percentage of biomass attributable to a single species-

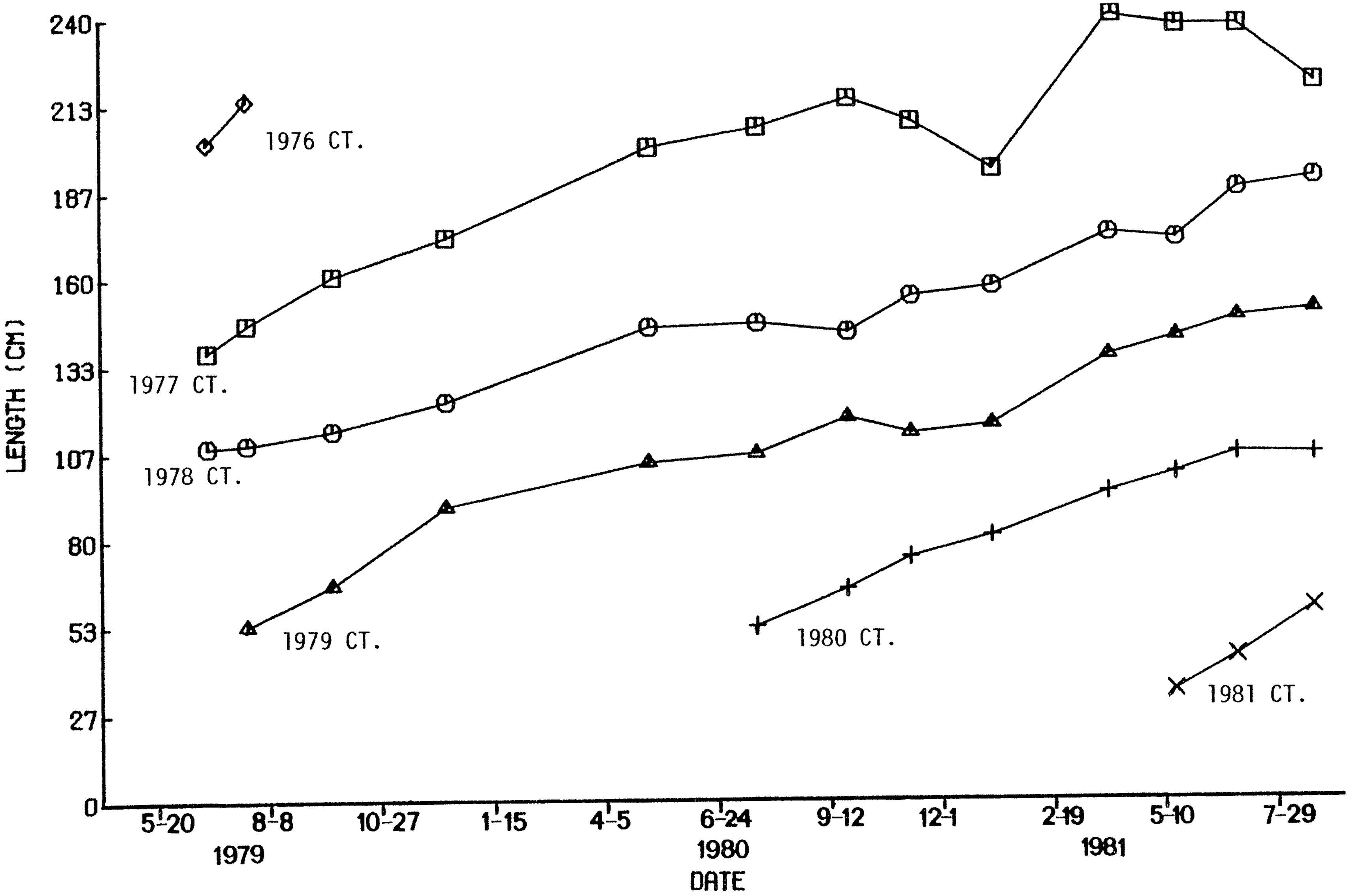


Figure 70. Mean lengths of the 1976 to 1981 year classes of cutthroat trout sampled in Bear Creek, June 1979 to August 1981.

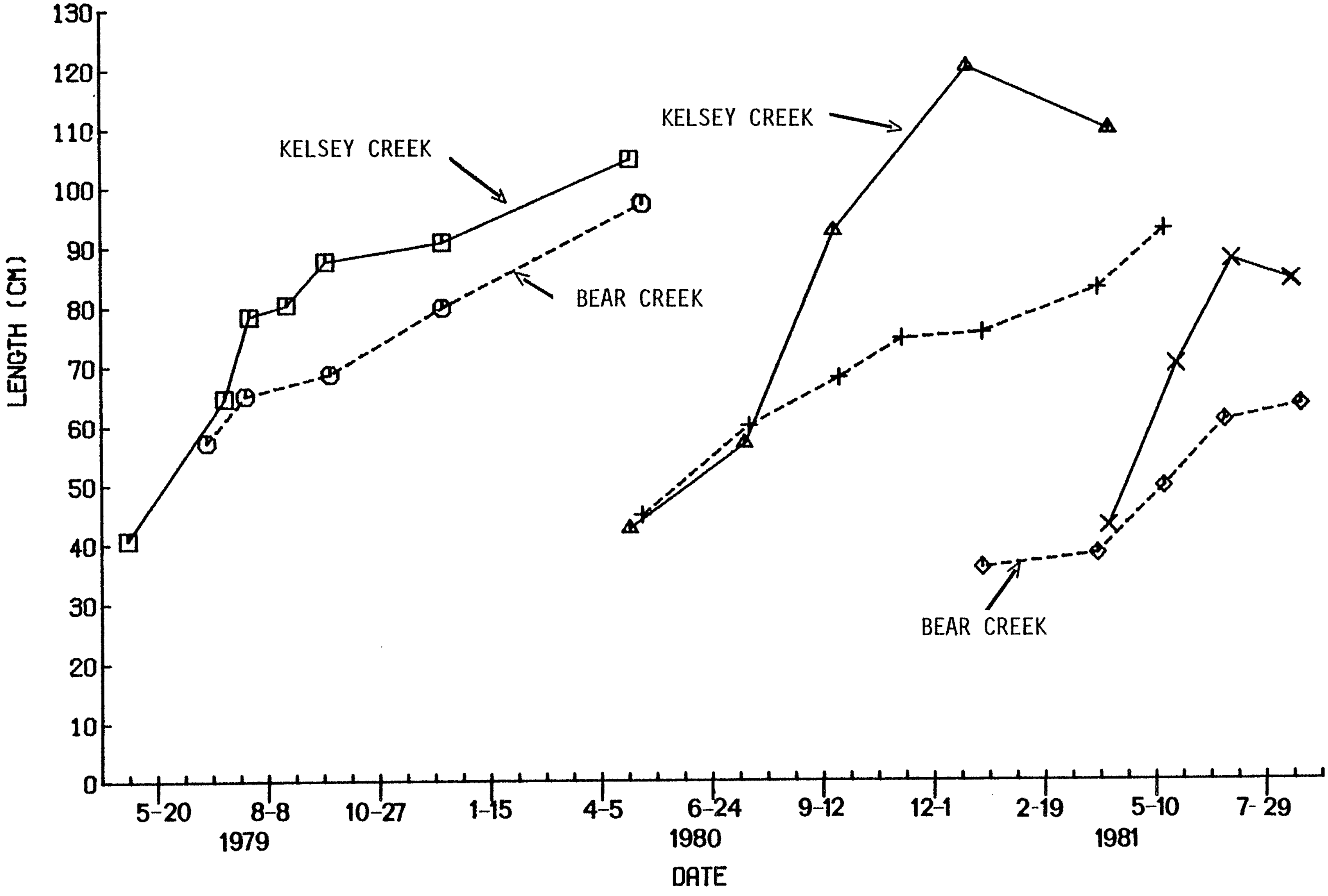


Figure 71. Mean lengths of the 1979, 1980, and 1981 year classes of coho salmon sampled in Bear and Kelsey Creeks.

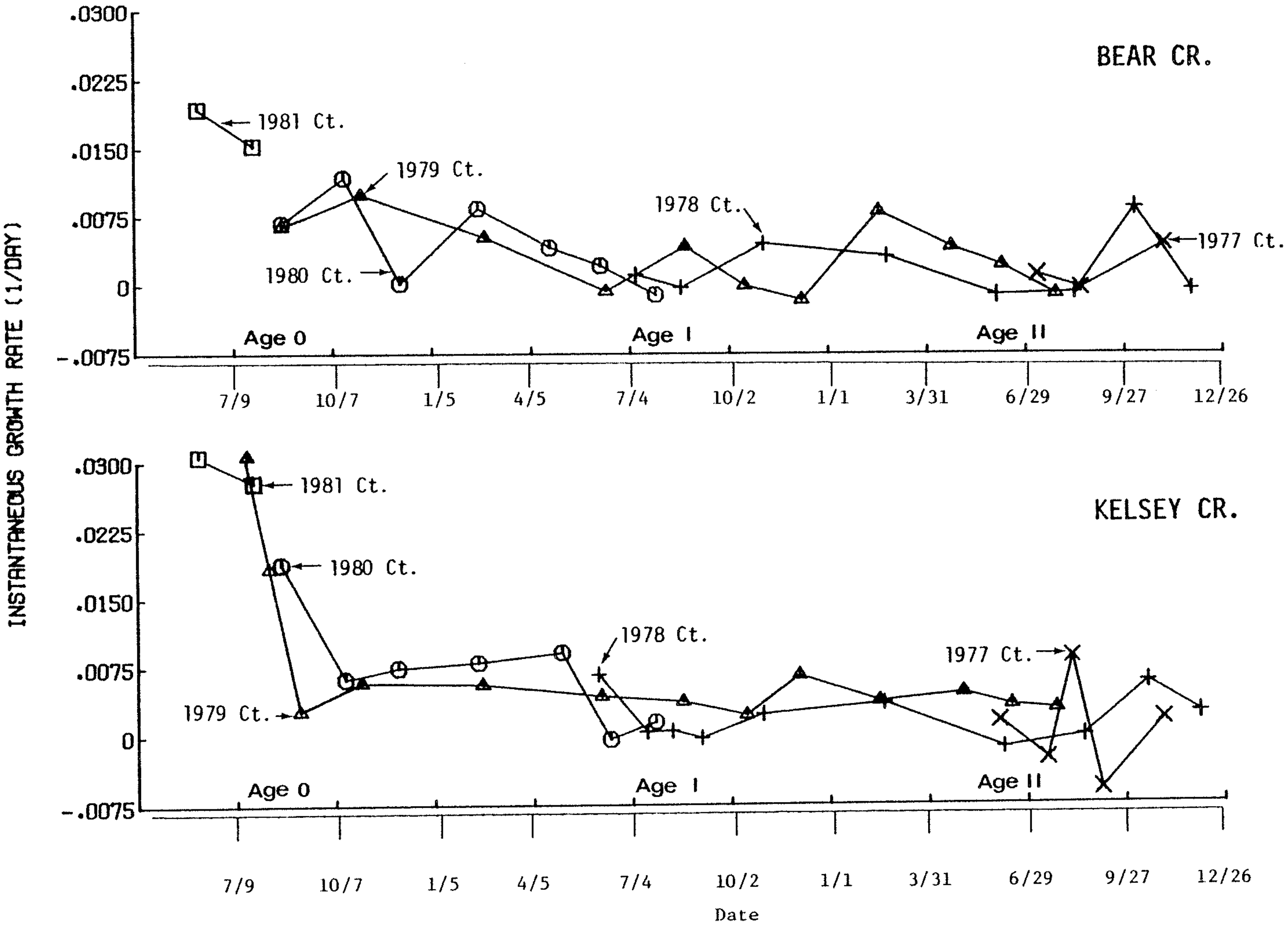


Figure 72. Instantaneous population growth (in weight) rates for the 1977 to 1981 year classes of cutthroat trout sampled in Bear and Kelsey Creeks, May 1979 to August 1981.

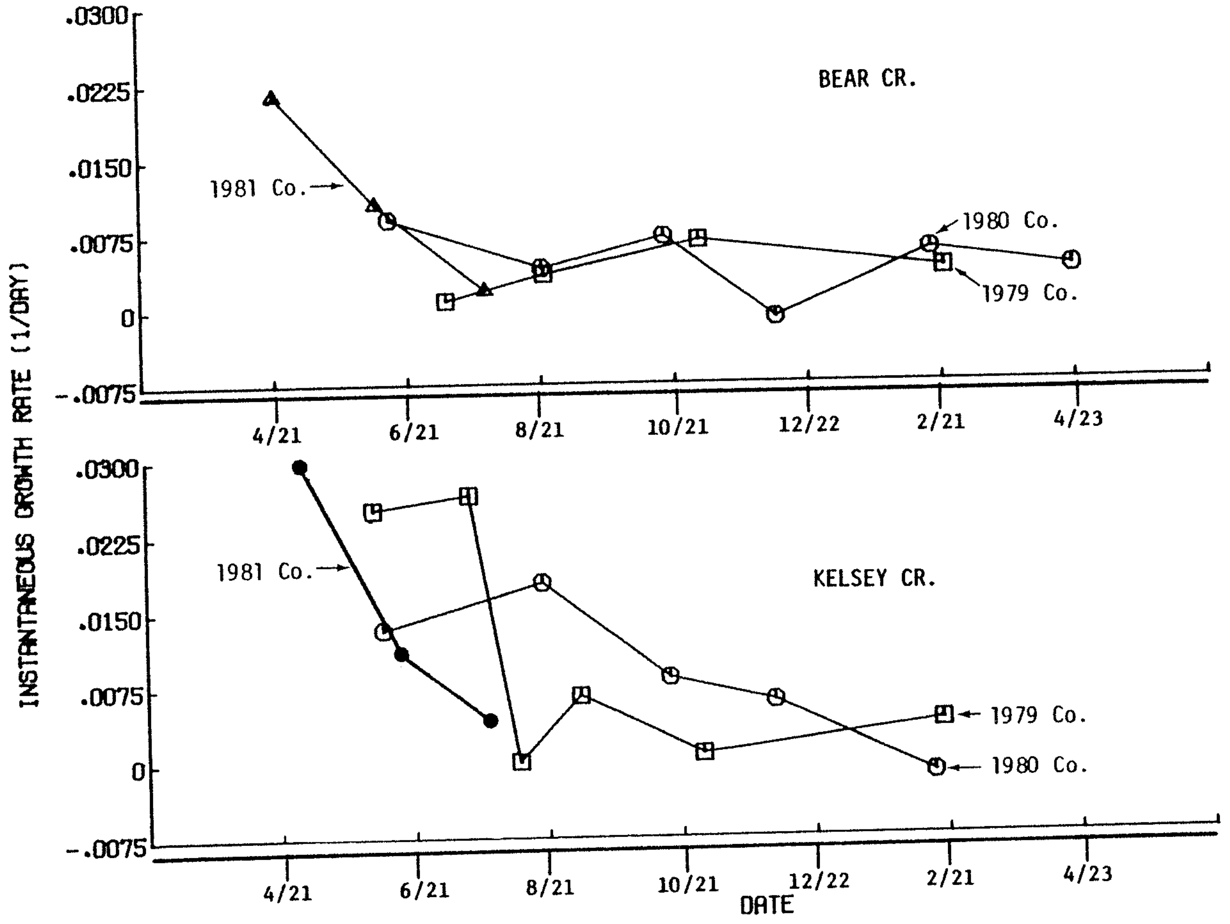


Figure 73. Instantaneous population growth (in weight) rates for the 1979-1981 year classes of coho salmon sampled in Bear and Kelsey Creeks.

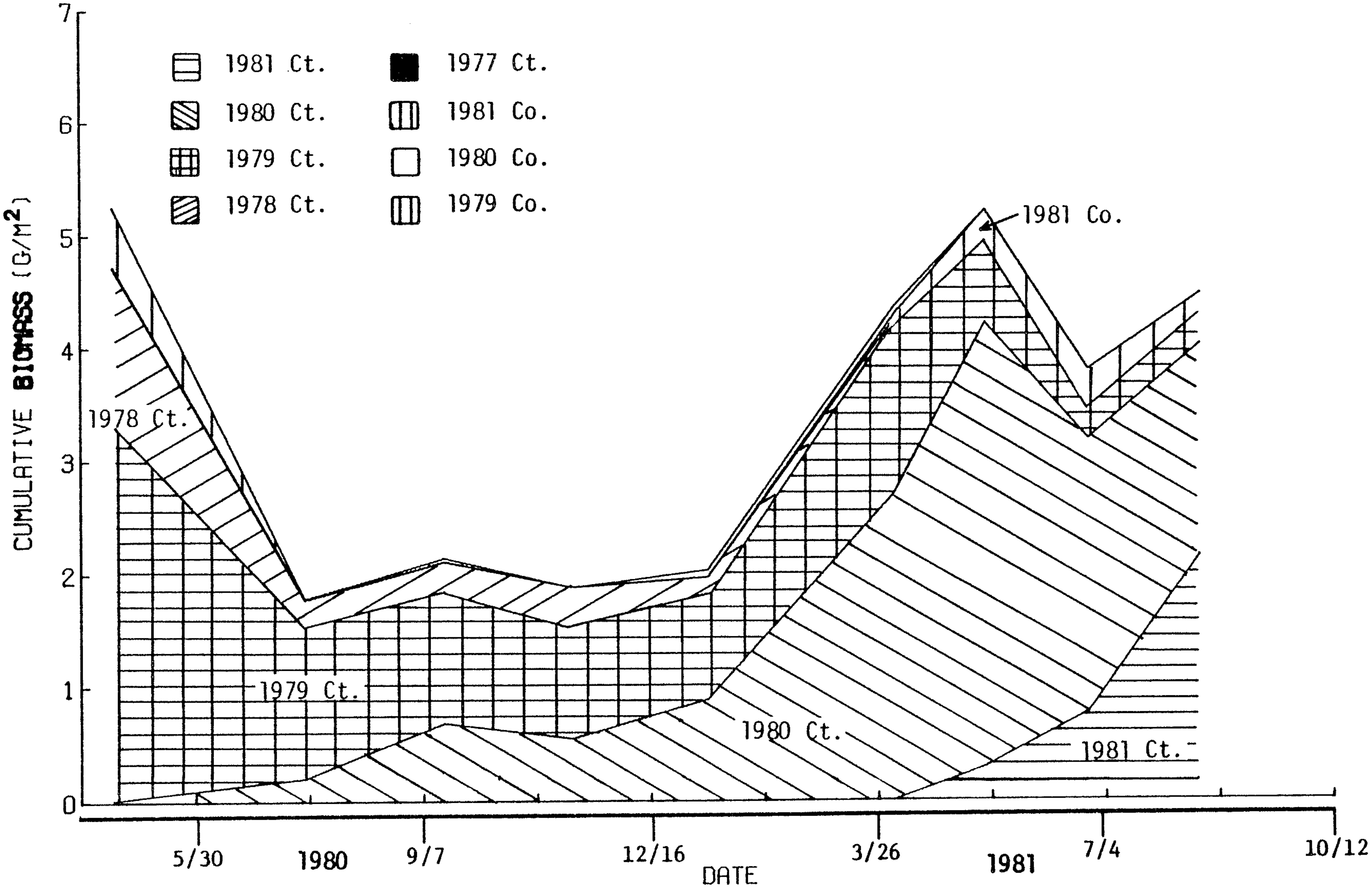


Figure 74. Mean biomass of salmonids, apportioned by species-age classes, at sampling sites in Kelsey Creek, April 1980 to August 1981.

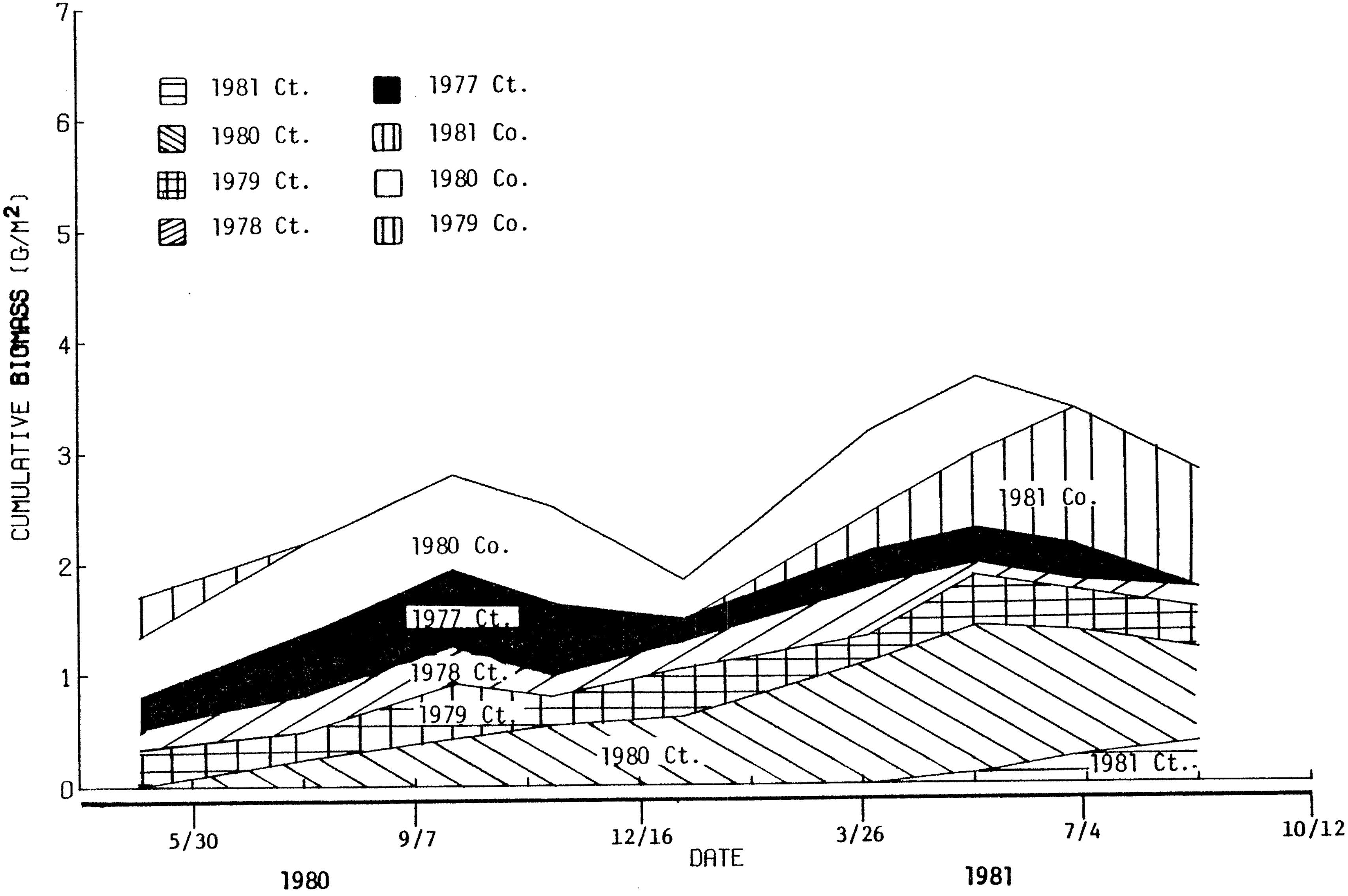


Figure 75. Mean biomass of salmonids, apportioned by species-age classes, at sampling sites in Bear Creek, April 1980 to August 1981.

age class was 30 percent, and the minimum was 12 percent. Although this difference was illustrated with data from 1980-81, it was apparent throughout the study period (Appendix Tables 4 and 5).

The biomass of salmonids supported by Kelsey Creek during the period April 1979 to August 1981 (Fig. 76) reflects the oscillations in salmonid abundance previously described. Biomass was relatively low (1.78 $g{\cdot}m^{-2}$) in the summer of 1980 in comparison to sampling dates before and after that time (maximum, 6.48 $g{\cdot}m^{-2}$). The two dominant seasonal trends, a rapid buildup of biomass in the late winter and early spring followed by a sharp decline in early summer, recurred on an annual basis in Kelsey Creek. The biomass of salmonids in Bear Creek has shown a generally increasing trend since December 1979 (Fig. 76) with a maximum of 3.7 $g{\cdot}m^{-2}$ in May of 1981.

7.5.5 Biomass of Nonsalmonids

The biomass of nonsalmonids supported by each stream was assessed only in August 1981. The derived length-weight regression equations, used to estimate the weight of fish not measured in the field, are shown for each species in Table 35.

Nonsalmonids were quite abundant in Bear Creek, with the various species of sculpins and dace contributing the majority of biomass. The average biomass of sculpins in Bear Creek was 1.49 $g{\cdot}m^{-2}$, and that of dace 0.76 $g{\cdot}m^{-2}$. Stickleback were of very limited importance, contributing only 0.03 $g{\cdot}m^{-2}$.

The biomass of nonsalmonids in Kelsey Creek averaged only 0.04 $g{\cdot}m^{-2}$, the majority (0.03 $g{\cdot}m^{-2}$) of which was accounted for by large-scale sucker. Dace, stickleback, and sculpin each contributed less than 0.005 $g{\cdot}m^{-2}$.

Earlier it was reported that the quantity of salmonid biomass supported

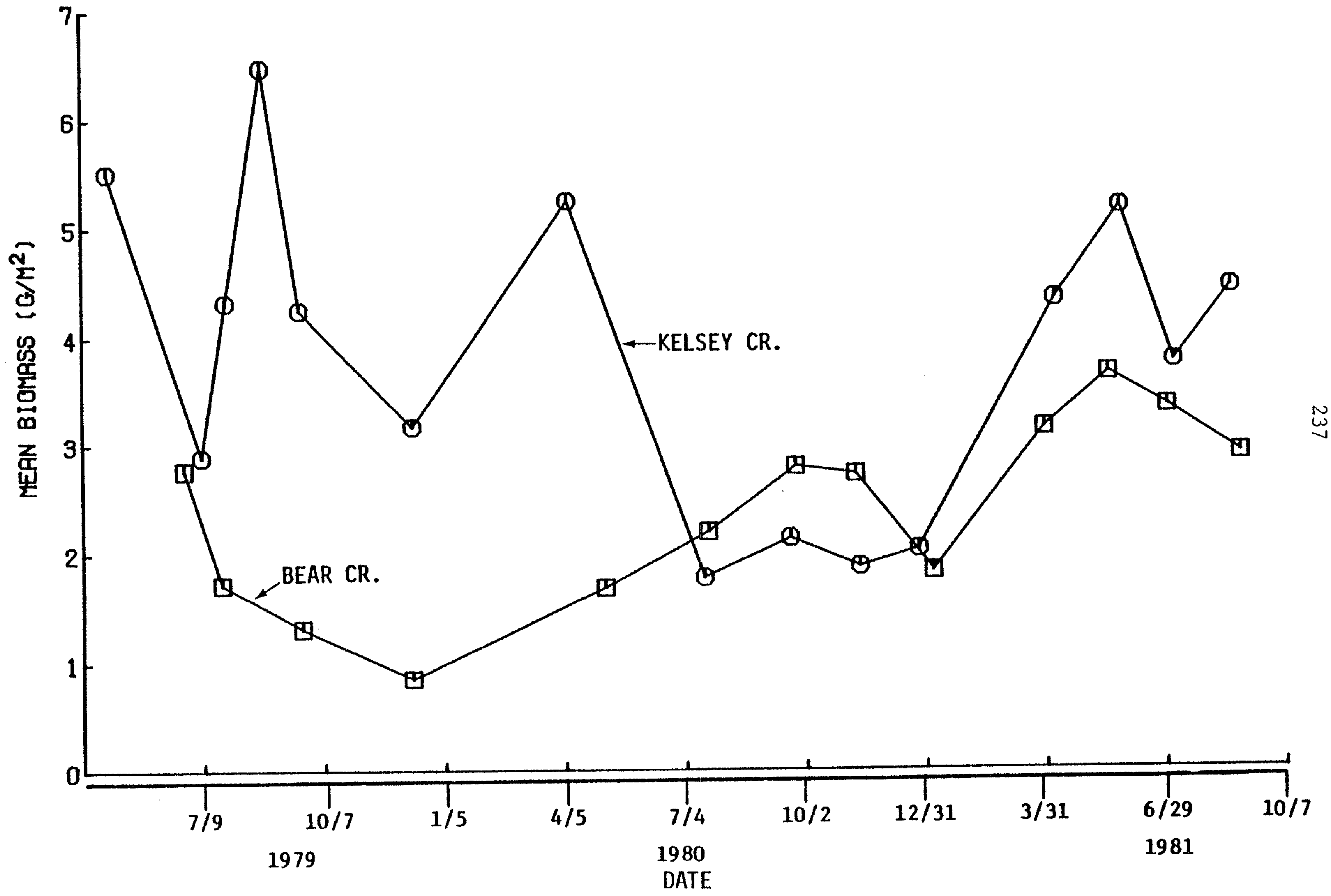


Figure 76. Mean biomass of salmonids at sampling sites in Bear and Kelsey Creeks, April 1979 to August 1981.

Table 35. Length-weight regression equations utilized to estimate the weight of nonsalmonids.

Stream	Species	n	R^2	Regression equation
Bear Creek	Longnose dace (*Rhinichthys cataractae*)	37	.97	$\ln(W) = -11/299 + 2.987 \ln(L)$
	Sculpin (*Cottus* sp.)	31	.69	$\ln(W) = -11/610 + 3.070 \ln(L)$
	Threespine stickleback (*Gasterosteus aculeatus*)	6	.95	$\ln(W) = -10.149 + 2.655 \ln(L)$
Kelsey Creek	Longnose dace (*Rhinichthys cataractae*)	20	.98	$\ln(W) = -12.349 + 3.205 \ln(L)$
	Sculpin (*Cottus* sp.)	13	.96	$\ln(W) = -13.447 + 3.508 \ln(L)$
	Threespine stickleback (*Gasterosteus aculeatus*)	6	.52	$\ln(W) = -11.263 + 2.973 \ln(L)$
	Largescale sucker (*Catastomus macrocheilus*)	13	.97	$\ln(W) = -12.254 + 3.176 \ln(L)$

by Bear Creek was somewhat less than that of Kelsey Creek in August 1981. However, when consideration was given to the biomass of nonsalmonids supported by each stream, it was found (Fig. 77) that it is actually Bear Creek which supports the greater quantity of total fish biomass (5.2 g m^{-2} versus 4.5 g m^{-2} in Kelsey Creek). It is also interesting to note that no single grouping of fish (coho, cutthroat, sculpins, etc.) accounts for more than 36 percent of the total biomass of fish present in Bear Creek.

7.5.6 Salmonid Production

Estimates of the interval production rates for the individual cutthroat and coho year classes censused during the study period are given in Appendix Tables 4 and 5. Production over the sample intervals varied in response to biomass and growth rates. Cutthroat production in Kelsey Creek was highest during the spring and summer months in all age groups (Fig. 78). Production rates were minimal and in some cases negative during the late summer period. As interval production rates generally decreased with increasing age, underyearling trout accounted for the majority of the total annual cutthroat production.

In Bear Creek, production was typically spread over a number of species-age groups (Fig. 79), and its seasonal pattern differed substantially from that observed in the urban stream. Among the most obvious differences was the upturn in salmonid production in Bear Creek during the fall months and the decline which followed it in early winter. As in Kelsey Creek, the spring and early summer months were a period of high productivity, but production in Bear Creek during mid-summer was low.

The general results cited above are illustrated in Figure 80, which is

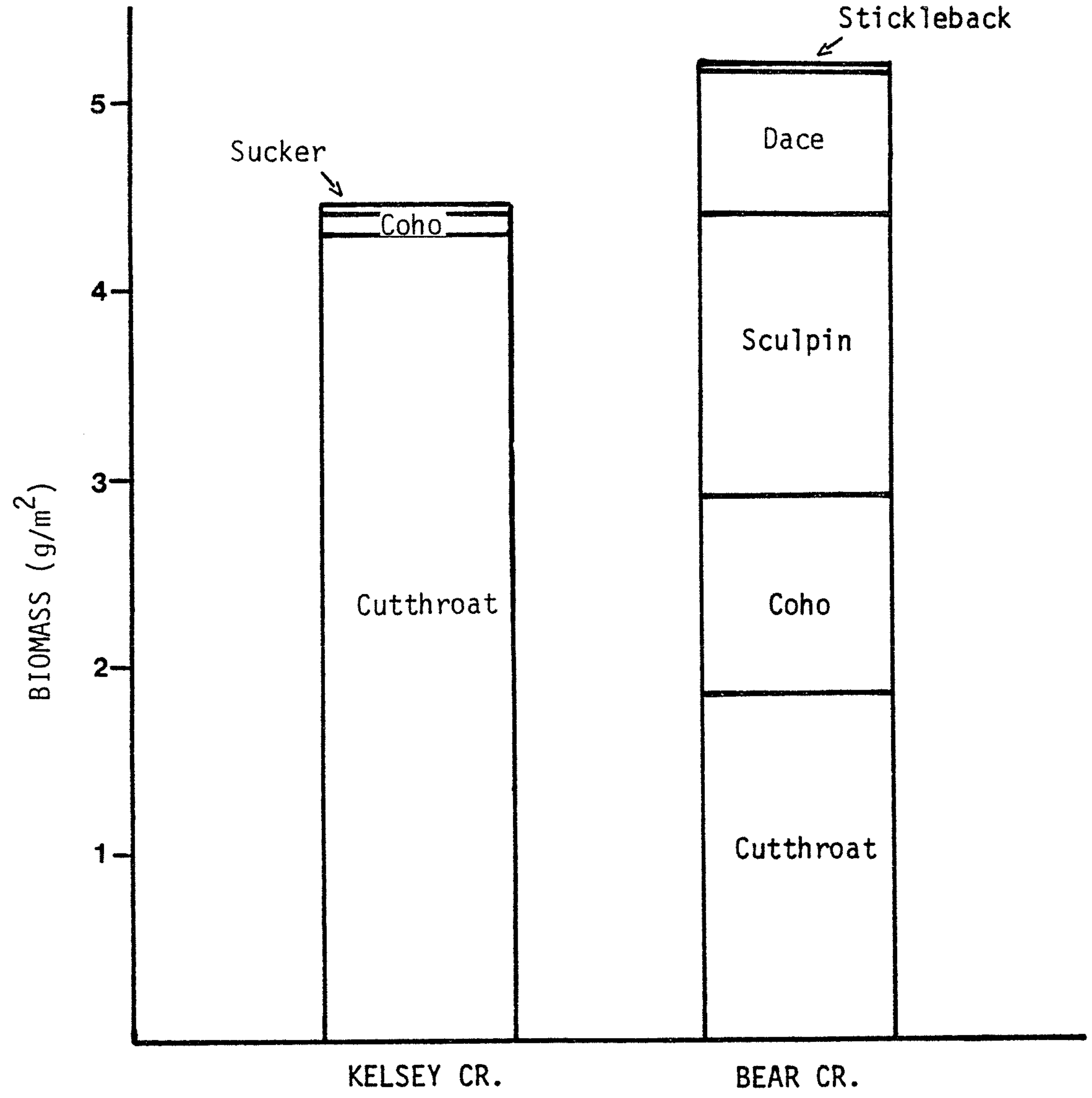


Figure 77. Average biomass of fish at sample sites in Bear and Kelsey Creeks, August 1981.

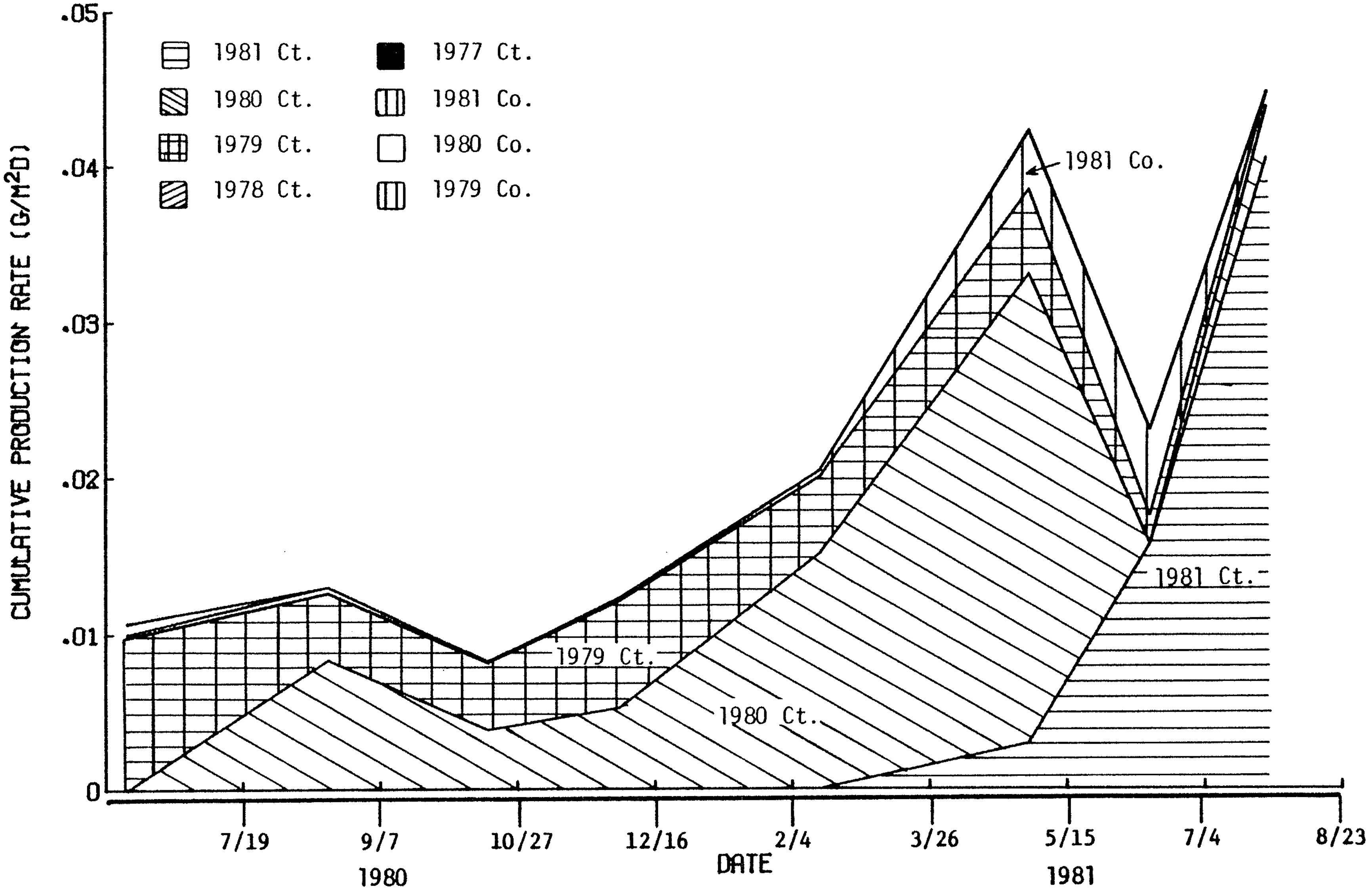


Figure 78. Mean production rate of salmonids, apportioned by species-age classes, at sampling sites in Kelsey Creek, April 1980 to August 1981.

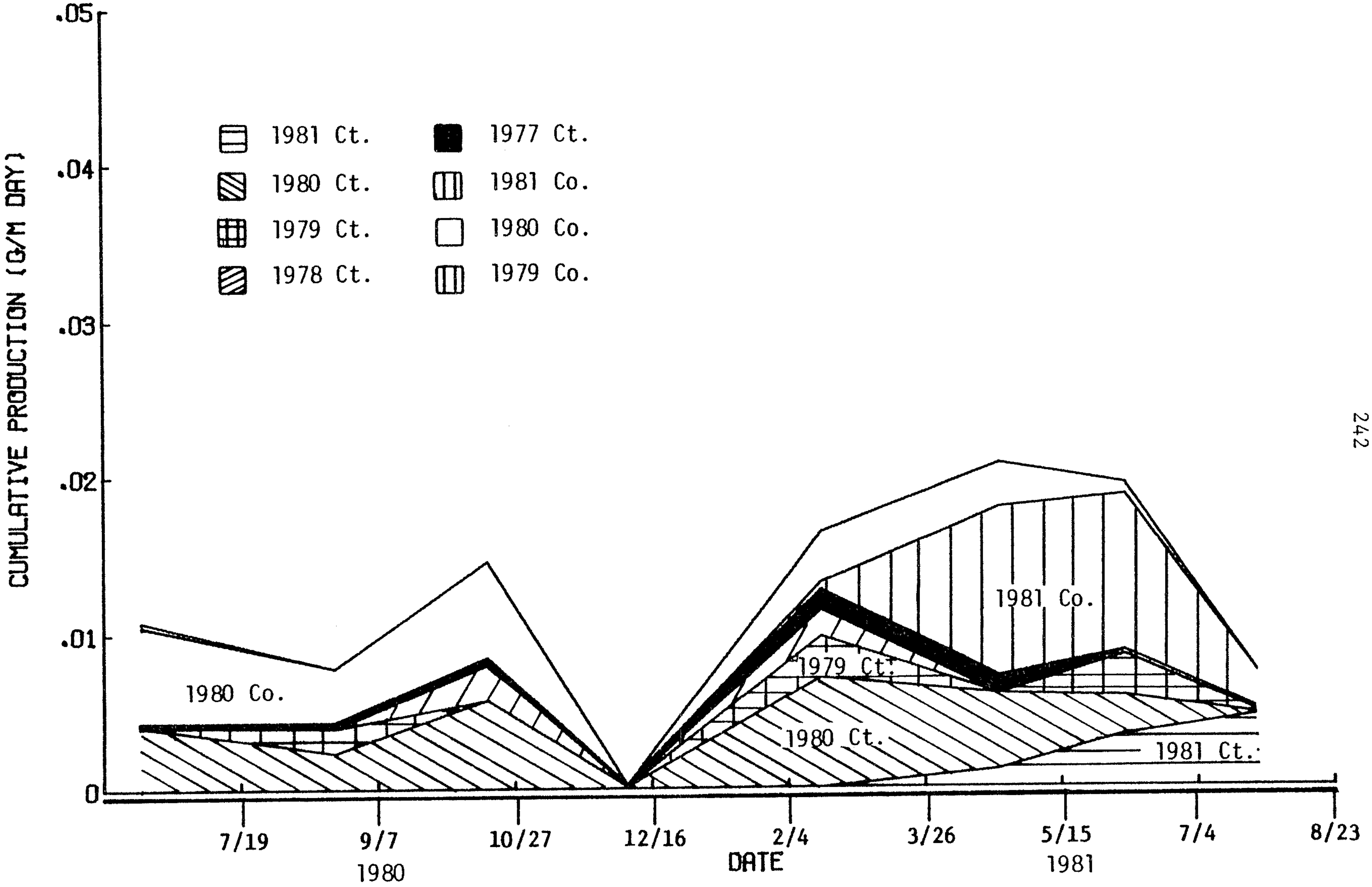


Figure 79. Mean production rate of salmonids, apportioned by species-age classes, at sampling sites in Bear Creek, April 1980 to August 1981.

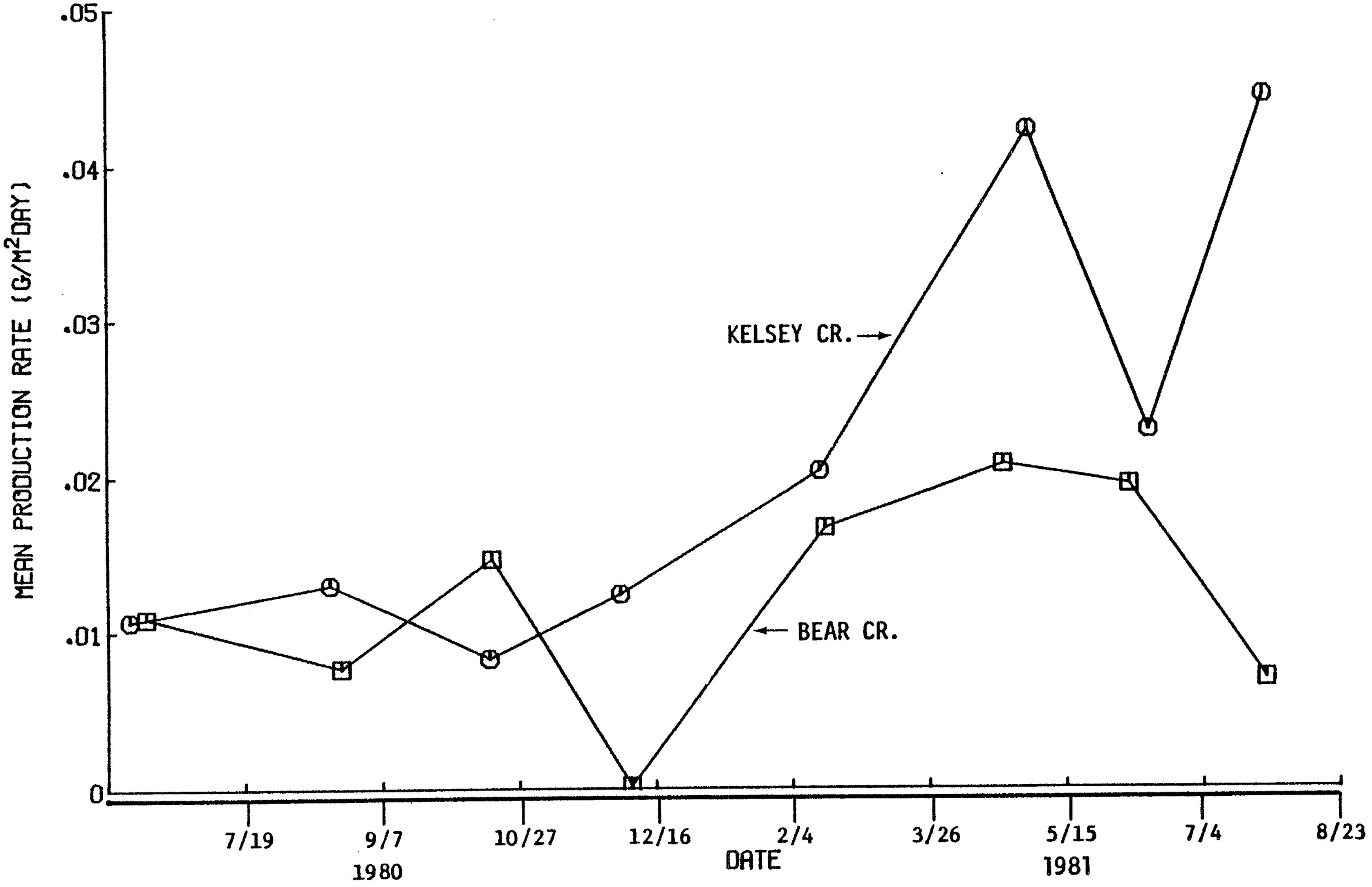


Figure 80. Mean production rate for salmonids in Bear and Kelsey Creeks, April 1980 to August 1981.

a plot of production in each of the streams in the period from April 1980 to August 1981. The total production of salmonids differed little between the streams from the commencement of sampling in April of 1980 to November of that year (Fig. 80). Thereafter, the total production of Kelsey Creek was considerably greater than that of Bear Creek. The difference between the two streams reached a maximum in the late summer of 1981, when the production per m^2 in Kelsey Creek was estimated to be over three times greater than that of Bear Creek.

The annual production by the major salmonid species-age groups varied little between 1979 and 1980 in Kelsey Creek (Table 36). In each year, age 0 cutthroat trout were responsible for a considerable portion of the annual production, contributing slightly more than 4 g m^{-2} per year. Production per age class of cutthroat trout declined consistently as age increased, and the net elaboration of tissue by coho salmon was less than that of any cutthroat trout age class. Bear Creek production in 1979 contrasted directly with that of Kelsey Creek (Table 37); coho salmon were the most productive species-age class, and age 0 cutthroat trout the least. This phenomenon was not repeated in 1980, when the strong 1980 year class of cutthroat trout dominated production.

The total annual salmonid production for Kelsey Creek for the years following July 23, 1979 and 1980, was estimated as 6.57 $g \cdot m^{-2}$ and 7.5 $g \cdot m^{-2}$, respectively. Annual salmonid production in Bear Creek during equivalent time periods was estimated as 1.94 $g \cdot m^{-2}$ and 4.6 $g \cdot m^{-2}$.

Turnover, or the ratio of annual production to mean biomass, is an additional means by which the growth rate of a population may be assessed. As calculated in this study, the turnover rate may be thought of as a weighted

Table 36. Annual production, mean biomass, and annual turnover for the predominant species-age classes of salmonids inhabiting Kelsey Creek, for the two years following July 26, 1979.

		Annual production ($g \cdot m^{-2}$)		Mean biomass ($g \cdot m^{-2}$)		Annual turnover	
Species	Age class	1979	1980	1979	1980	1979	1980
Cutthroat trout							
	Age 0 → I	4.0	4.1	1.9	1.7	2.1	2.4
	Age I → II	1.1	1.5	1.4	1.0	0.8	1.5
	Age II → III	0.3	0.1	0.7	0.2	0.5	0.6
Coho salmon		0.2[1]	0.1[2]	0.3	<0.1	0.7	1.1

---

[1] Period extends from July 26, 1979 through April 26, 1980.

[2] Period extends from July 26, 1980 through May 25, 1981.

Table 37. Annual production, mean biomass, and annual turnover for the predominant species-age classes of salmonids inhabiting Bear Creek, for the two years following July 26, 1979.

Species	Age class	Annual production ($g \cdot m^{-2}$)		Mean biomass ($g \cdot m^{-2}$)		Annual turnover	
		1979	1980	1979	1980	1979	1980
Cutthroat trout							
	Age 0 → I	0.1	1.4	0.1	0.8	1.3	1.8
	Age I → II	0.1	0.4	0.2	0.4	0.7	1.2
	Age II → III	0.3	0.3	0.3	0.3	0.9	1.0
Coho salmon	Age 0 → I	1.3[1]	0.9[2]	0.5	0.7	1.5	1.4

[1] Period extends from July 26, 1979 through June 1, 1980.

[2] Period extends from July 26, 1980 through June 18, 1981.

mean of the instantaneous growth rates for each sample interval, where the weighting terms are a function of the mean biomass on that period. Turnover rates in Kelsey Creek (Table 36) were greater in 1980 than in 1979 for all species-age groups, and in each year they declined as the age of cutthroat trout increased. Turnover rates for age 0 and I cutthroat trout were also greater in Bear Creek in 1980 relative to 1979, but those of coho salmon and age II cutthroat varied only slightly between the two years (Table 37). It is interesting to note that the rate of turnover of age 0 cutthroat was greater in Kelsey Creek than Bear Creek, but the reverse condition applied for age 0 coho salmon.

7.5.7 Age and Year Class Comparisons

A stable cutthroat age structure in Kelsey Creek is implied by the relative abundance of fish in each cohort: younger fish are more numerous than older ones, and the number of fish in all year classes decreased during the study period. Additionally, there is strong evidence that the density of age i + 1 trout present in 1980 is a function of age i abundance in the previous year (Table 38). The population size of the 1979 year class in July 1979 is significantly correlated with September 1979 and April 1980 densities of the same year class. Similarly, the density and biomass estimates of age 0 fish in September was highly correlated with estimates of age I fish in the following April. Comparisons of density and biomass of the 1979 year class trout in April 1979-80 also resulted in significant correlations. A negative correlation was found between 1978 year class abundance in July 1979 and April 1980. Biomass correlations for the same year class were significant for the July 1979-April 1980 and July 1979-July 1980 comparisons.

Table 38. Correlation between a) densities ($No \cdot m^{-2}$) and b) biomass ($g \cdot m^{-2}$) of age group i in 1979 and 1980 in Kelsey Creek. Regression coefficient = r; sample size = 5 stations.

		Density		Biomass	
		r	Probability	r	Probability
		Cutthroat trout			
April 1979 - April 1980	Age 1	0.88	0.05	0.82	NS
	Age 2	-0.41	NS	-0.41	NS
July 1979 - July 1980	Age 0	0.56	NS	0.28	NS
	Age 1	-0.92	0.05	-0.49	NS
	Age 2	0.38	NS	0.01	NS
		Coho salmon			
April 1979 - April 1980	Age 0	-0.20	NS	-0.34	NS
	Age 1	0.60	NS	0.25	NS
July 1979 - July 1980	Age 0		NS		NS

Correlations of the density of age I trout in 1978 and 1980 (Table 39) were positive for April and negative for July comparisons. All other age group correlations were insignificant.

When all of the age groups are combined and a correlation coefficient is calculated, the density relationship between years is significant for both April and July comparisons. Combining age groups has the effect of increasing the sample size but assumes that mortality rates are similar between age groups. Since this assumption does not appear to be valid, the results of the combined age group comparisons are probably inconsequential.

There is little evidence of a stable age structure in the Bear Creek cutthroat population, although the detection of a significant relationship between year class strength in 1979 and 1980, and the comparisons of age group density and biomass in both years, was hampered by small sample sizes ($n = 3$ stations). Density estimates of the 1977-1979 cutthroat year classes at time t and time $t + 1$ were only correlated in the July-September comparison of the 1977 year class (Table 40). None of the paired biomass comparisons were significant.

No correlations were found between the local densities of age 0, I, and II trout in the July samples in 1979-80 from Bear Creek (Table 41). Correlation coefficients, however, were generally high ($r > 0.94$) which in view of the small sample size suggests a tendency towards correlation. The coefficient b (3.190) of the July 1979-80 age 0 regression was greater than any other slope calculated for Kelsey and Bear Creek age group comparisons, indicating a large increase in the abundance of young-of-the-year trout in Bear Creek in 1980 over the previous year. This conclusion is corroborated by the sharp increase in age 0 trout biomass ($b = 2.895$) in the July 1980 samples;

Table 39. Correlation between a) densities ($No \cdot m^{-2}$) and b) biomass ($g \cdot m^{-2}$) of year class i at time t and time t + 1 in Kelsey Creek. Correlation coefficient = r; t-value = t; sample size = 5 stations.

time t - time t + 1	Year class	Density		Biomass	
		r	Probability	r	Probability
		Cutthroat trout			
April 1979 - April 1980	1977	.93	0.05	.93	0.05
	1978	-.45	NS	.48	NS
	1979	Not present in April 1979			
July 1979 - September 1979	1977	.74	NS	.69	NS
	1978	.82	NS	.49	NS
	1979	.93	0.05	.84	NS
July 1979 - April 1980	1977	-.12	NS	-.15	NS
	1978	-.97	0.05	.90	0.05
	1979	.94	0.05	.84	NS
July 1979 - July 1980	1977	Not present in July 1980			
	1978	-.48	NS	-.94	0.05
	1979	-.50	NS	-.41	NS
September 1979 - April 1980	1977	.21	NS	.31	NS
	1978	-.65	NS	-.48	NS
	1979	.95	0.05	.92	0.05
		Coho salmon			
May 1979 - April 1980	1979	.62	NS	.28	NS
July 1979 (early) - September 1979	1979	.35	NS	.32	NS
July 1979 (late) - September 1979	1979	.75	NS	.85	NS
September 1979 - April 1980	1979	.94	0.05	.69	NS

Table 40. Correlation between a) densities (No·m^{-2}) and b) biomass (g·m^{-2}) of year class i at time t and time t + 1 in Bear Creek. Correlation coefficient = r; sample size = 3 stations.

time t - time t + 1	Year class	Density		Biomass	
		r	Probability	r	Probability
		Cutthroat trout			
July 1979 - September 1979	1977	1.00	0.05	0.99	NS
	1978	0.53	NS	0.28	NS
	1979	0.99	NS	0.99	NS
July 1979 - May 1980	1977	0.19	NS	0.35	NS
	1978	0.46	NS	0.13	NS
	1979	0.92	NS	0.95	NS
July 1979 - July 1980	1977	0.81	NS	0.84	NS
	1978	0.94	NS	0.95	NS
	1979	0.95	NS	0.94	NS
September 1979 - May 1980	1977	-0.90	NS	-0.89	NS
	1978	-0.50	NS	-0.91	NS
	1979	0.87	NS	0.90	NS
		Coho salmon			
June 1979 - September 1979	1979	1.00	0.05	1.00	0.05
June 1979 - May 1980	1979	0.64	NS	0.28	NS
September 1979 - May 1980	1979	0.60	NS	0.32	NS

Table 41. Correlation between a) densities ($No \cdot m^{-2}$) and b) biomass ($g \cdot m^{-2}$) of age group i in 1979 and 1980 in Bear Creek. Regression coefficient = r, sample size = 3 stations.

		Density		Biomass	
		r	Probability	r	Probability
		Cutthroat trout			
July 1979 - July 1980	Age 0	0.99	NS	1.0	0.01
	Age 1	0.97	NS	0.96	NS
	Age 2	0.96	NS	0.95	NS
		Coho salmon			
July 1979 - July 1980	Age 0	0.77	NS	1.0	0.05

the correlation coefficient was highly significant.

A long-term residency of coho salmon in the study areas of Kelsey Creek is implied by the correlation of local densities of the 1979 year class in September 1979 and April 1980. In Bear Creek the abundance (and biomass) of the same year class was correlated between June and September samples in 1979.

The biomass of age 0 coho in July of 1979 and 1980 was positively correlated.

7.5.8 Fish Health

Laboratory examination of cutthroat trout afflicted with respiratory anomalies revealed that three general types of lesions were present in the gill filaments. These lesions included hyperplasia of the respiratory epithelium of the lamellae (Plate 2), bifurcation of filaments (Plate 3), and telangiectasis of the lamellar capillaries (Plate 4). However, no pathogens were observed in Giemsa-stained sections of gill tissue. Externally, these conditions were manifested in a visible swelling of gill tissue beyond the operculum.

The percentage of fish sampled in Kelsey Creek afflicted with respiratory anomalies varied from 0 to 77 percent in response to seasonal and spatial factors, as well as the age and species of fish under consideration. The effects of age, season, and species are most easily examined if the data from all sites are pooled for a given date and species-age group. Focusing initially on the 1981 year class of cutthroat (Fig. 81), it may be seen that the proportion afflicted increased rapidly after mid-May. The proportion of age I and II cutthroat afflicted similarly increased from July to August (Fig. 81), after declining steadily throughout the spring. Also notable in Fig. 81 is

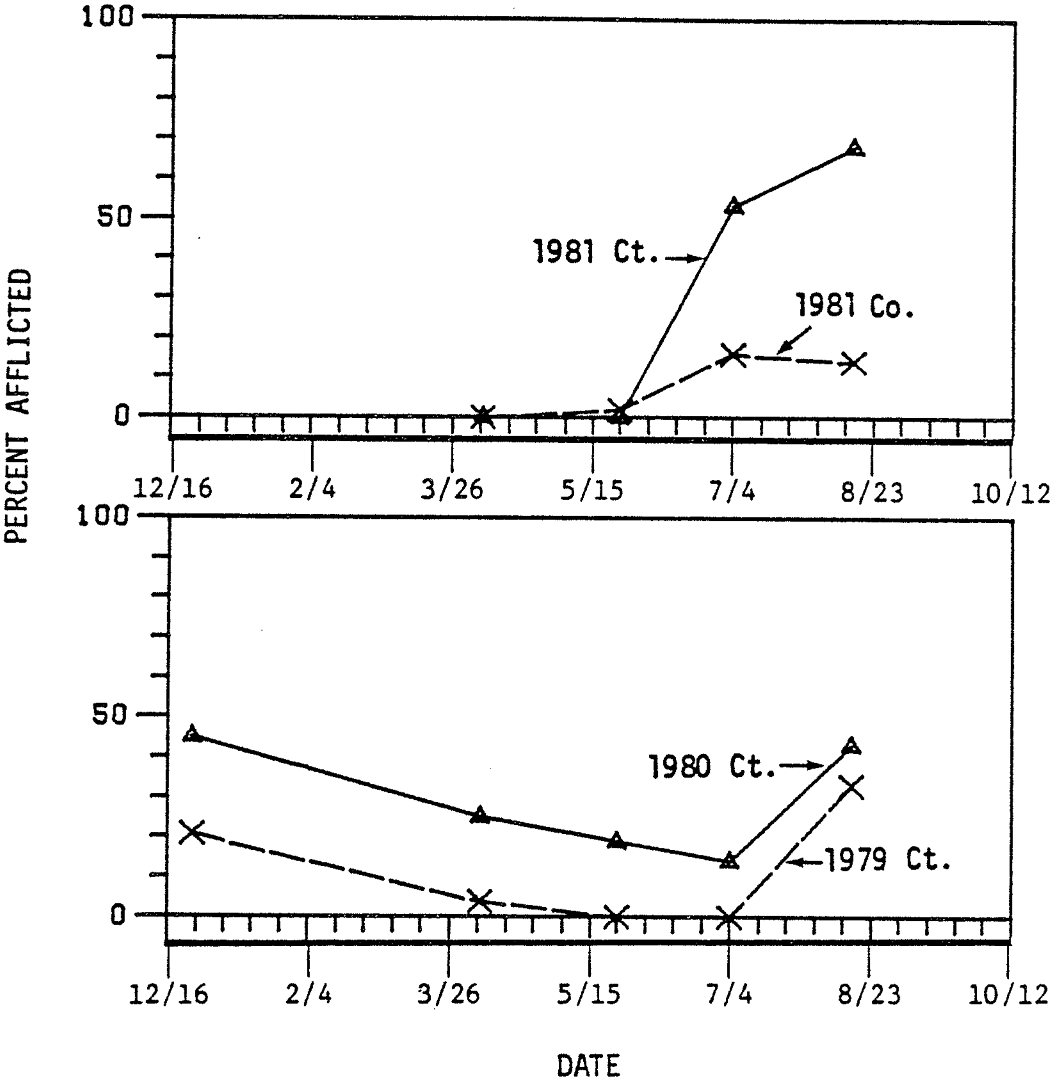


Figure 81. Percentage of the 1981 cohort of coho salmon and the 1979, 1980, and 1981 cohorts of cutthroat trout sampled in Kelsey Creek displaying damaged gills, by sampling date.

Plate 1. Normal gill filament from a cutthroat trout captured in Kelsey Creek. (Magnification = 200X)

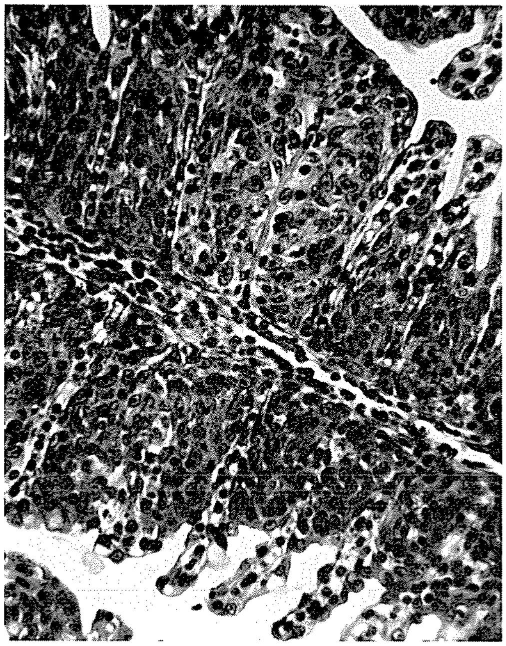


Plate 2. Gill filament from a cutthroat trout captured in Kelsey Creek exhibiting hyperplasia of the respiratory epithelium. (Magnification = 200X)

Plate 3. Bifurcation of gill filaments of a cutthroat trout captured in Kelsey Creek. (Magnification = 200X)

Plate 4. Gill filament exhibiting telangiectasis of the lamellar capillaries. (Magnification = 200X)

the progressive reduction with age in the percentage of cutthroat displaying gill damage, and the apparent relative immunity of juvenile coho to this condition.

Spatially, the incidence of damaged gills on members of the 1980 (Fig. 82) and 1981 (Fig. 83) year classes of cutthroat trout generally decreased as sampling proceeded downstream. This is particularly evident in the August 1981 samples, which occurred at a time when fish migration was minimal. Chi-square tests were employed to establish homogeneous subsets among the Kelsey Creek sites. For all 1981 sample dates, the incidence of affliction for the 1980 year class of cutthroat trout was significantly ($\alpha \leq .05$) lower at sites KC4 and KC5 (Table 42) than at KC1. The incidence of damaged gills was also lowest ($\alpha \leq .05$) at KC5 (Table 42) for the 1981 cutthroat trout year class.

No cutthroat trout and only two of the coho salmon sampled in Bear Creek were observed to have damaged gills. Chi-square tests between the Bear and Kelsey Creek subsets for differences in the proportion of fish afflicted with respiratory anomalies were frequently significant. The incidence of gill-damaged cutthroat trout of the 1980 year class was significantly ($\alpha \leq .05$ greater at all sites on Kelsey Creek for all sample dates but July (Table 42). Similarly, for the 1981 year class of cutthroat trout, the incidence of gill damage at all sites on Kelsey Creek was significantly greater than in Bear Creek in July and August (Table 43). Due to the limited sample sizes for 1981 year class coho and 1979 cutthroat, all sites on Kelsey Creek were pooled without statistically testing for homogeneity. Utilizing this pooled data, the percentage of the 1981 year class of coho afflicted with respiratory anomalies was significantly greater in the urban stream in July and August. Sample sizes for the 1979 year class of cutthroat trout were limited, and

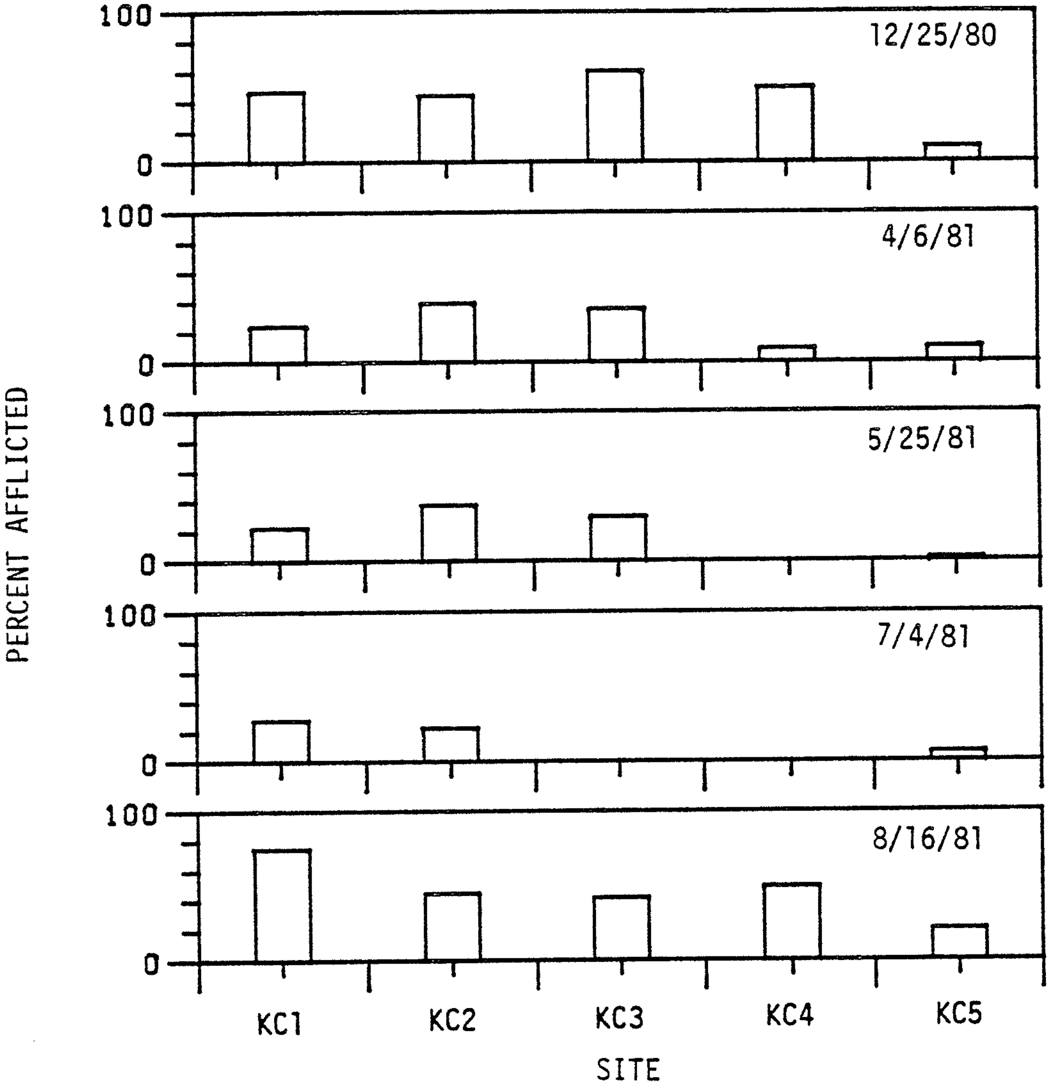


Figure 82. Percentage of the 1980 cohort of cutthroat trout sampled in Kelsey Creek displaying damaged gills, by site and date.

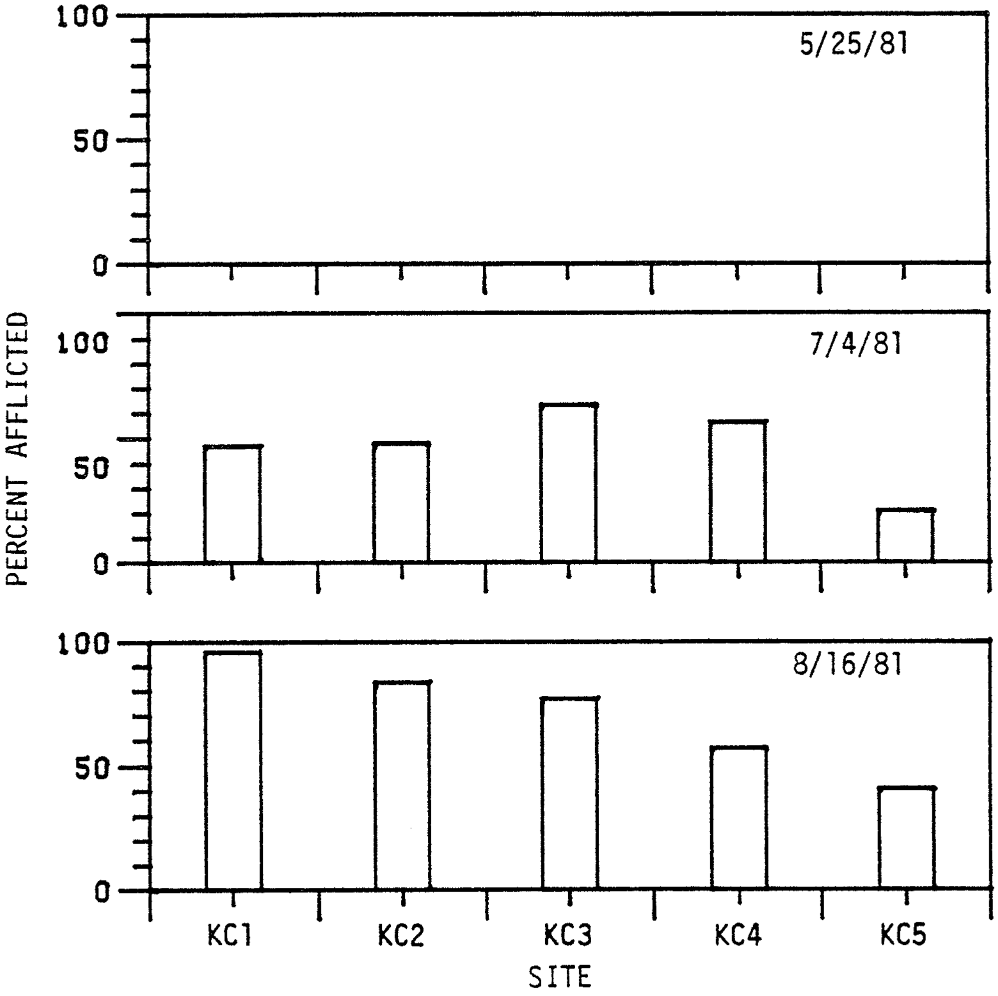


Figure 83. Percentage of the 1981 cohort of cutthroat trout sampled in Kelsey Creek displaying damaged gills, by site and date.

Table 42. Results of chi-square tests for the percentage of the 1980 year class of cutthroat trout affected with damaged gills ($\alpha = .05$ for all tests).

	Test hypothesis		
Date	Homogeneity within Kelsey Creek	Homogeneity within Bear Creek	Homogeneity between Kelsey and Bear Creeks
12/80	Accepted	Accepted	Rejected BC1, BC2, BC3 < KC1, KC2, KC3, KC4, KC5
4/81	Rejected KC4, KC5^{1} < KC1, KC2, KC3	Accepted	Rejected BC1, BC2, BC3 $<^{2}$ KC4, KC5 KC1, KC2, KC3
5/81	Rejected KC4, KC5^{1} < KC1, KC2, KC3	Accepted	Rejected BC1, BC2, BC3, KC4, KC5 < KC1, KC2, KC3
7/81	Rejected KC3, KC4, KC5^{1} < KC1, KC2	Accepted	Rejected BC1, BC2, BC3, KC3, KC4, KC5^{3} < KC1, KC2^{2}
8/81	Rejected KC2, KC3, KC4, KC5^{2} < KC1	Accepted	Rejected BC1, BC2, BC3^{3} < KC2, KC3, KC4, KC5 $<^{2}$ KC1

[1] Subset pooled without testing due to small sample size.

[2] Expected frequencies less than 5 in these subsets.

[3] Expected frequency less than 1.

Table 43. Results of chi-square tests for the percentage of the 1981 year class of cutthroat trout affected with damaged gills ($\alpha = .05$ for all tests).

	Test hypothesis		
Date	Homogeneity within Kelsey Creek	Homogeneity within Bear Creek	Homogeneity between Kelsey and Bear Creeks
5/81	Accepted	Accepted	Accepted
7/81	Rejected KC5 KC1, KC2, KC4 KC3	Accepted	Rejected BC1, BC2, BC3 <[1] KC5 < KC1, KC2, KC4 < KC3
8/81	Rejected KC5 KC4 KC1, KC2, KC3	Accepted	Rejected BC1, BC2, BC3 < KC5 < KC4 < KC1, KC2, KC3

[1] Expected frequency less than 5.

only the fish sampled in December differed significantly between the streams.

7.5.9 Salmonid Populations in Kelsey Creek Tributaries

The fish populations of the three principal tributaries of Kelsey Creek were assessed once in August 1981. Nonsalmonid species collected were longnose dace (Richards Creek and West Tributary), sculpin (Richards Creek), and large-scale sucker (West Tributary). Of the salmonid species, cutthroat trout were the most prevalent, appearing in samples from each of these streams. In addition, coho salmon were found in Richard and Valley Creeks, and a single sockeye salmon was captured in West Tributary.

The density of salmonids supported by each stream varied greatly, with a range extending from a minimum of .11 fish/m^2 in West Tributary to a maximum of 2.93 fish/m^2 in Valley Creek (Table 44). A major reason for the relatively dense salmonid population of Valley Creek was its large coho salmon population. The density of coho salmon in Valley Creek was more than 25 times greater than that in Richards Creek, and more than 50 times greater than that in Kelsey Creek. However, salmonids inhabiting Valley Creek were significantly smaller in length than those from the other tributary streams (Figs. 84 and 85).

The percentage of cutthroat trout sampled which displayed anomalies in their respiratory apparatus exhibited an inverse relationship to fish density (Table 45). Where sample sizes were sufficient, the percentage of cutthroat trout afflicted in Richards and Valley Creeks was always far less than in Kelsey Creek or West Tributary. No coho salmon in either Valley or Richards Creeks were found to have damaged gills.

Table 44. Density of salmonids in Kelsey Creek, Valley Creek, Richards Creek, and West Tributary, in late August, 1981.

Stream	Coho Age 0 No./m^2	Cutthroat Age 0 No./m^2	Cutthroat Age I No./m^2	Cutthroat Age II No./m^2	Salmonid total No./m^2
Valley Cr.	1.03	1.53	.33	.04	2.93
Richards Cr.	.04	.65	.14	.04	.87
Kelsey Cr.[1]	.02	.51	.06	<.01	.60
West Trib.[2]	0	.10	.01	0	.11

[1] Mean of 5 sites.

[2] Mean of 2 sites.

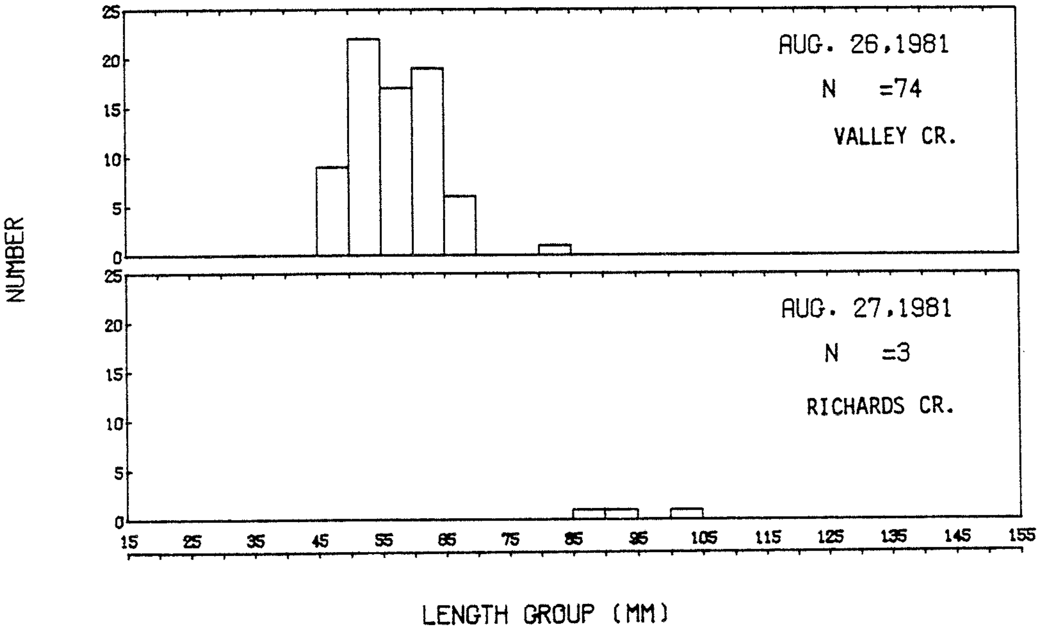


Figure 84. Length frequency histograms for coho salmon sampled in Valley and Richards Creeks, August 1981.

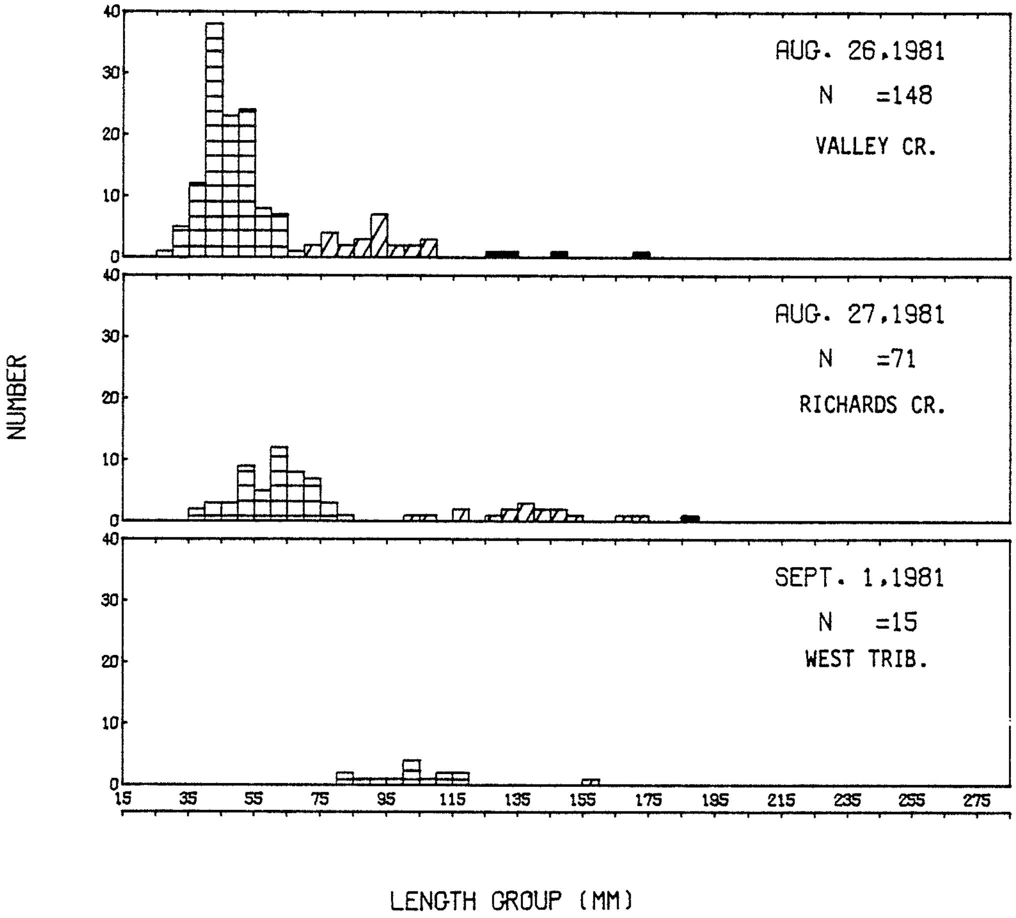


Figure 85. Length frequency histograms for cutthroat trout sampled in Valley Creek, Richards Creek, and the West Tributary, August 1981.

Table 45. Incidence of gill damage on salmonids captured in Kelsey Creek, Valley Creek, Richards Creek, and the West Tributary, in late August 1981.

Stream	Coho Age 0 % afflicted	Cutthroat Age 0 % afflicted	Cutthroat Age I+ % afflicted
Valley Cr.	0	6	10
Richards Cr.	0	5	19
Kelsey Cr.	14	68	43
West Trib.	—	50	(0)[1]

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[1] Sample of age 1.

8.0 DISCUSSION

8.1 Substrate Quality

The measurement of particle size distribution of spawning bed materials has been used repeatedly in the past to assess the quality of the intragravel environment for developing salmonid eggs and larvae. Salmonids have been observed to spawn in substrates of varying composition; however, fisheries biologists have long recognized that only a narrow range in the proportion of fine particles seems to be tolerated.

Studies conducted in Pacific Northwest watersheds, outside of Oregon, in which human impact has been slight, have found fine sediment levels ranging from 3.1 to 20.6 percent (wet volume), with an overall mean of 11.1 percent (Table 46). The geology of the Oregon coastal range is apparently composed of more erosive marine sediments which contribute higher levels of fine sediment to associated streams (Moring and Lantz 1974); for this reason, studies conducted in Oregon coastal streams have been excluded from Table 46. The proportion of sediment (wet volume) less than .841 mm was 22 percent in Kelsey Creek and 18 percent in Bear Creek. Thus while the streambeds in most watersheds will have significantly fewer fine particles than found in the study streams, the levels observed in both Kelsey and Bear Creeks are within the range observed in undisturbed watersheds.

The level of fines in the samples appears to be a more sensitive measure of substrate quality than does the mean geometric diameter of the particle size distribution. Dissolved oxygen concentrations in intragravel water are more highly correlated with percent fines than with d_g values, implying that the percentage of fines in the substrate has a more pronounced effect on

Table 46. Percentage of fine sediment (wet volume) in streambed core samples from Pacific Northwest streams not impacted by development.

Stream (state)	Number of samples	Cutoff point	Percent fines	Source
Upstream Harris R. (AK)	25	<.833	13.9	McNeil and Ahnell (1960)
Anan Cr. (AK)	5	<.833	5.7	"
Upper Clearwater R. (WA)	27	<.850	8.3	Cederholm and Salo (1979)
S. Fork Hoh R. Trib. (WA)	6	<.850	3.1	"
S. Fork Hoh R. (WA)	19	<.850	8.3	"
Tshlecthy Cr. (WA)	6	<.850	6.0	"
Bob Cr. (WA)	6	<.850	4.9	"
Harlow Cr. (WA)	5	<.850	9.6	"
Stequaleho Cr. No. 1 (WA)	43	<.850	6.9	"
Sollecks R. Nos. 1-3	91	<.850	7.9	"
Little Lost Man Cr. (CA)	20	<.833	16.3	Woods (1980)
Bummer Lake Cr. (CA)	21	<.800	10.2	Burns (1972)
S. Fork Yager Cr. (CA)	10	<.800	16.4	"
Little N. Fork Noyo R. (CA)	27	<.800	20.0	"
S. Fork Casper Cr. (CA)	20	<.800	20.6	"
			Mean 11.1	

gravel porosity, permeability, and water exchange. In general, variations in the proportion of fine particles have been a more tractable means of distinguishing differences between sampling dates, individual sampling areas, and experimental streams. Percent fines is also more closely associated with in-channel hydraulic variables than is d_g.

Streambed composition is highly variable in both time and space. The natural heterogeneity of the substrate is acted upon by local changes in water flow and sediment transport conditions. A quantitative relationship between hydraulic variables and features of the bed composition in Kelsey and Bear Creeks has been demonstrated, explaining in part the appreciable variation in composition between locations within the two streams. In particular, an increase in the hydraulic gradient of an area is accompanied by an increase in the mean geometric diameter of the streambed materials. A correlation between bed slope and substrate composition has been documented previously in streams of comparable drainage area and discharge (Scullion and Milner 1980). As the gradient of a channel normally decreases in a downstream direction (Leopold et al. 1964), a progressive downstream reduction in the mean particle size was expected. That it was not observed indicates that this type of trend may be discernable only over distances greater than the lengths of stream sampled in this study.

Variation in average discharge over the substrate is also associated with changes in bed composition. Discharge may be expected to influence the local hydraulic conditions of flow (e.g., water depth and velocity) and, consequently, the deposition, intrusion, and flushing of fine sediments from the gravel. The composition of the substrate in both Kelsey and Bear Creeks was notably coarser during winter periods of high average flows than at other times of the year. The increase in percent fines (and decrease in d_g) observed during March, June, and September is associated with changes in discharge and possibly sediment supply. Seasonal differences in the rate of sediment loading to drainage channels have been reported in several studies (Guy 1963, Walling and Gregory 1970). An increase in construction activity, changes in soil structure, and greater intervals between flood events during the summer months are factors which may influence the bed composition in the study streams.

8.2 Streambed Stability

The dynamic equilibrium concept of stream channels (Leopold et al. 1964) is a basic tenet of modern geomorphology. According to this hypothesis, channel characteristics such as width, depth, slope, and bed composition form a self-regulating system whose components are in quasi-equilibrium with sediment inputs and the discharge regime. Channel adjustments may be expected to occur in watersheds in which any of these entities are disturbed.

The frequency and depth of scour in Kelsey Creek indicate that the dynamic increase in peak flows induced by urban development within the watershed has shifted the channel equilibrium point. In contrast, in the control stream, which was exposed to similar climatological conditions, a stable relationship appears to exist, as evidenced by the limited scour which occurred.

The model developed to estimate the scour-induced mortality of incubating salmonid embryos in Kelsey Creek may be verified to a limited degree by reviewing the analysis of previous research presented in Section 4.3. A flood with a recurrence interval of 5 years in Kelsey Creek (1970-1980 data) has a peak instantaneous discharge of about 11 m^3/s, and would be expected to increase coho embryo mortality by about 20 percent. In terms of the small stream-large stream dichotomy, Kelsey Creek most closely resembles the former, in which the scour mortality inflicted by a 5-year flood would be expected to be 10 percent or less.

8.3 Intragravel Water Quality

Sensitivity of salmonid embryos and alevins to changes in pH is dependent on the stage of development (Daye and Garside 1977, Daye 1980); however, the pH levels encountered in this study are well within the tolerances of all salmonid life stages. The intragravel pH, ranging from 6.5 to 7.6, did not differ significantly between streams on any given date and tended to decline during the spring months.

Lower autumnal temperature and pH levels contributed to the reduction in un-ionized ammonia concentrations observed, since NH_3-N decreases with decreasing temperature and alkalinity (Sawyer and McCarty 1978). The concentration of un-ionized ammonia, however, is primarily dependent on total ammonia concentration, which, in turn, depends on the oxidation of available organic nitrogen. Total and un-ionized ammonia levels are significantly higher in Kelsey Creek than in Bear Creek. The un-ionized form is not present in sufficient concentrations to cause mortality of salmonid eggs. Rankin (1979) estimated that total mortality due to a free ammonia level of 0.023 mg $NH_3\text{-N}\cdot l^{-1}$ to be 9 percent. This concentration was an order of magnitude higher than the maximum level recorded in the interstitial water samples from Kelsey Creek.

Interstitial dissolved lead concentrations were quite low in each stream on the single date which they were sampled. The maximum concentration observed, 6 μg/l, is well within the 1-10 μg/l range typically observed in natural rivers and lakes throughout the world (Livingstone 1963). The excellent survival of embryos in the streamside bioassay also indicates that surface water quality in Kelsey Creek has not been sufficiently degraded to reduce survival.

Intragravel oxygen concentrations in disturbed and control watersheds have frequently been assessed (Ringler and Hall 1975, Woods 1980, Tagart 1976, Turnpenny and Williams 1980, Hall and Lantz 1969). The mean monthly oxygen concentrations found in these studies are presented in Figure 86. Figure 86 also shows the intragravel oxygen concentrations in Kelsey Creek for the periods of December 1979 to May 1980 and December 1980 to May 1981. Oxygen concentrations in Kelsey Creek are far below those characteristic of undisturbed watersheds. During the period from December to May, the oxygen concentrations are near the mean observed in disturbed watersheds as a whole. This is perhaps the strongest indication that the intragravel environment in Kelsey Creek has been impacted by urban developments within the watershed. This analysis also suggests that spawning bed conditions in Bear Creek may be deteriorating. During the winter and spring months, intragravel oxygen conditions appear to be intermediate between those characteristic of disturbed and undisturbed watersheds (Fig. 86).

8.4 Salmonid Embryo Bioassays

The instream embryo bioassays utilized a new procedure to study the effects of alterations in the intragravel environment on the survival of embryos. Results from the bioassays indicate that when oxygen concentrations are measured by the standpipe technique, embryo mortality will be high when the mean dissolved oxygen concentration drops below 6.5-7.5 mg/l. This conclusion is corroborated by previous research, which suggests a threshold condition

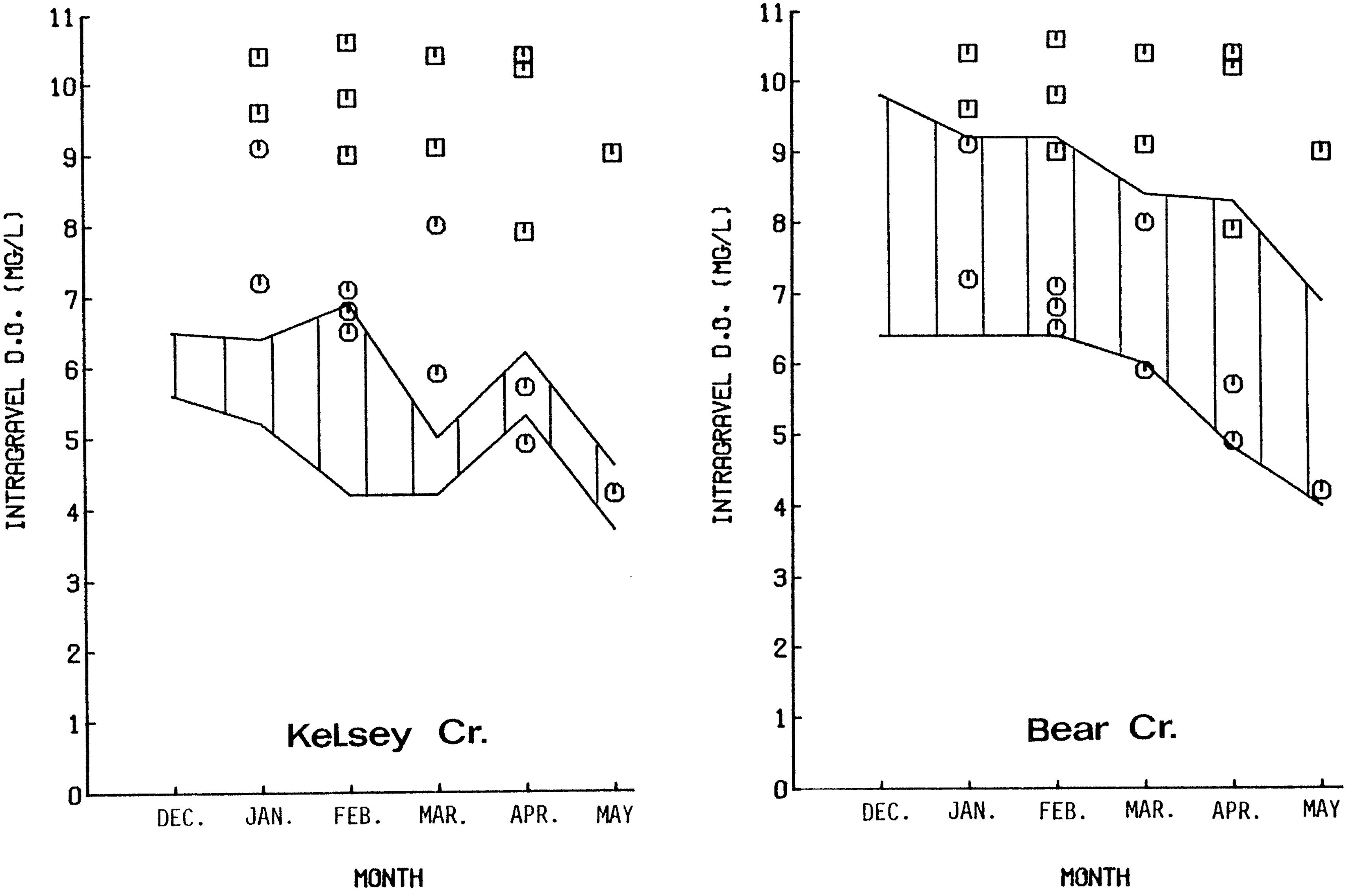

Figure 86. Average monthly intragravel dissolved oxygen concentrations in disturbed (○) and control (□) watersheds in comparison to those of Kelsey and Bear Creeks (shaded areas) during the 1979-80 and 1980-81 winter-spring incubation periods (see text for sources).

exists in the 7-8 mg/l range (Fig. 5). Thus the sensitivity of the methodology utilized in this study to determine embryo mortality induced by intragravel dissolved oxygen deficiencies appears to be approximately equal to that obtained from the Vibert box technique.

The survival of embryos during the winter bioassay was significantly greater in Bear Creek than in Kelsey, but no difference in survival existed in the spring bioassay. Earlier researchers in Alaska who employed the SEPD (White 1980) estimated the survival of planted eggs (eyed) to be equal to or greater than that of natural spawners. If indeed eyed eggs planted by the SEPD experience mortality similar to that of eggs in natural redds, then the instream bioassay results lend credence to the observation that dissolved oxygen conditions in both the urban and control streams have been impacted by urban development. However, until further testing of the SEPD is completed, these results should be interpreted cautiously.

8.5 Population Characteristics

8.5.1 Outmigration

A potential source of error in the estimation of the number of outmigrant cutthroat is the lack of effort during daylight hours in the period from November through March. However, numerous authors (Meehan and Siniff 1962, Crawford et al. 1967, Hoar 1958) have indicated that salmonids typically migrate during the night, and this appears to be the case in Kelsey Creek as well, at least prior to the onset of the primary migration period. Fishing the net during daylight on November 27 and December 13 failed to yield a single cutthroat.

Perhaps the best measure of the relative "health" of a stream in the

Pacific Northwest is the number of smolts which it produces. The number of salmonid smolts produced per low flow surface area of Kelsey Creek is approximately 40 percent less than has been observed in other studies (Table 47). It is important to note, however, that the ratio of coho salmon to trout (cutthroat and rainbow-steelhead) smolts in Kelsey is nearly one to one, while in other streams the production of coho smolts typically vastly exceed that of trout (Table 48). Why the Kelsey Creek watershed is so sparsely populated with coho salmon will be discussed in Section 6.5.2; at this point it is sufficient to say that the relative abundance of cutthroat trout leads to a reasonable explanation for the apparent poor salmonid smolt production of the watershed. As reviewed and elaborated upon by Allen (1969), research has indicated that a definite relationship exists between the length of salmonids and their territorial requirements. In fact, Allen's data suggest that the territorial requirements of an average coho smolt from Kelsey Creek would be less than 20 percent of those of a typical cutthroat smolt. For this reason the smolt production of the Kelsey Creek watershed may be appropriate for a stream in which cutthroat trout are relatively abundant.

Unfortunately, determining the smolt production of the Kelsey Creek watershed as a whole may inaccurately portray the condition of Kelsey Creek itself. Two tributary streams, Valley and Richards Creeks, have been significantly less impacted by urbanization than the mainstem, and together they might tend to raise the average production for the Kelsey Creek watershed as a whole. Electrofishing conducted throughout the watershed in August 1981 substantiates this hypothesis.

If the length of smolts at the time of outmigration may be used as a rough measure of the quality of the environment in which they were reared,

Table 47. Numbers of coho salmon and trout smolts (cutthroat and rainbow-steelhead) produced per square meter of low flow stream surface area for streams in the Pacific Northwest as well as Kelsey Creek.

Stream (Locality)		Coho smolts M^2	Trout smolts M^2	Total smolts M^2	Source
Little Stawamus River					Armstrong and Argue (1977a)
(British Columbia)	1974	.57	.03	.60	
	1975	.50	.07	.57	
Meign Creek					"
(British Columbia)	1974	.60	.01	.60	
	1975	.27	<.01	.27	
Pastuch Creek					Armstrong and Argue (1977b)
(British Columbia)	1975	.41	.02	.43	
Deer Creek					Chapman (1965); Lowry (1965)
(Oregon)	1960	.40	.12	.52	
	1961	.49	.13	.61	
	1962	.59	.10	.68	
Flynn Creek					"
(Oregon)	1960	.32	.29	.61	
	1961	.28	.17	.45	
	1962	.51	.14	.65	
Needle Branch					"
(Oregon)	1960	.42	.18	.61	
	1961	.18	.12	.30	
	1962	.52	.06	.58	
Mean		.43	.10	.53	
Kelsey Creek	1981	.17	.15	.31	

Table 48. Ratio of coho to trout smolts (cutthroat and rainbow-steelhead) produced by Pacific Northwest streams as well as Kelsey Creek.

Stream (Locality)	Years	No. coho smolts / No. trout smolts	Source
Little Stawamus River (B.C.)	1974-75	13.1	Armstrong and Argue (1977a)
Meighn Creek (B.C.)	1974-75	30.0	"
Pastuch Creek (B.C.)	1975	20.5	Armstrong and Argue (1977b)
Lymn Creek (B.C.)	1972	60.0	Mason (1974)
Big Beef Creek (WA)	1978-80	19.3	Seiler, Neuhauser, and Ackley (1981)
Deer Creek (OR)	1960-62	4.3	Chapman (1965); Lowry (1965)
Flynn Creek (OR)	1960-62	2.1	"
Needle Branch (OR)	1960-62	4.2	"
	Mean	19.2	
Kelsey Creek	1981	1.1	

then the Kelsey Creek watershed may be classified to be of good to excellent quality. Kelsey Creek cutthroat appear to have grown considerably more rapidly than cutthroat in streams where downstream trapping has previously been conducted (Table 49). In particular, the average length of age I cutthroat smolts in Kelsey Creek is close to the average length of age II cutthroat smolts in other streams, and, similarly, the length of age II smolts in Kelsey Creek is more typical of the previously reported lengths for age III smolts. In contrast, the average length of coho sampled at the downstream net is similar to that of smolts produced in other streams throughout the Pacific Northwest (Table 50).

The discrepancy in growth rates between the two species may be related to differences between the specific streams in which they reared. As noted above, the cutthroat were very abundant in the mainstem of Kelsey Creek, but coho were rarely observed.

There is considerable evidence in the literature that fish must reach a certain critical size before smolting may begin (Hoar 1976, Johnston and Eales 1970, Elson 1957). For this reason, it is not surprising that the age at which cutthroat migrate from Kelsey Creek is one to two years earlier than in streams where accelerated growth has not occurred (Table 50). The majority of fish in Kelsey Creek outmigrate at age I, with a considerable portion outmigrating at age II. In comparison, cutthroat smolts from other streams are generally age II-IV. Researchers at many of the streams listed in Table 49 (Deer Creek, Flynn Creek, Needle Branch, Gobar Creek, and Bear Creek) have suggested that most, if not all, age I fish passing downstream may spend an additional year in fresh water (below the trap site) before entering the ocean. It might be reasonable, then, to believe that the cutthroat migrating from Kelsey Creek

Table 49. Age composition, length, and run timing of outmigrant cutthroat trout in Pacific Northwest streams.

Stream (state)	Age I $\bar{L}$	Age I % run	Age II $\bar{L}$	Age II % run	Age III $\bar{L}$	Age III % run	Age IV $\bar{L}$	Age IV % run	Age V $\bar{L}$	Age V % run	Date of peak outmigration I	II	III	IV	All	Period of operation
Sand Creek (OR)[1]																
1947	106.7	5.1	132.1	28.0	175.3	42.3	210.8	20.6	238.8	4.0						Year round
Deer Creek[2]	94.3	12.0	131.7	48.3	168.1	34.0	209.7	5.7								Year round
Flynn Creek		16.7		66.2		15.7		3.4								
Needle Branch (OR)		42.4		39.6		11.1		6.9								
1963																
Gobar Creek (WA)																
1977[3]															4/28	3/25-6/10
1978[4]	105.6	19.7		28.5		45.6		5.2							5/11	3/16-5/29
1979[5]		7.5	151.0	63.8	174.0	28.7									5/25	3/19-6/01
Bear Creek (WA)[6]																
1977	77.5	13.2	95.1	60.4	128.3	14.3	157.0	1.1							5/14	4/20-9/25
Minter Creek (WA)[7]																
1953															∿4/24	Year round
Kelsey Creek																
1981	134.1	57.0	174.9	41.9	229.0	1.1					4/15	4/21	2/17		4/15	Year round
Lymn Creek (B.C.)[8]																
1972	92.0	35.6	137.2	57.8	172.1	6.6					4/04	4/11	4/16, 4/24		4/04	4/04-9/26

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[1] Summer (1962). Average lengths computed from Table 7, length-frequencies by age (25.4 mm groupings).

[2] Lowry (1965). Average lengths computed from Table 1, length-frequencies by age (25 mm groupings).

[3] Crawford et al. (1978).

[4] Crawford, Seider and Chilcote (1979).

[5] Chilcote, Seider and Crawford (1980).

[6] June (1981). Migration timing dates from Fig. 15.

[7] Salo and Bayliff (1958). Data from Fig. 27.

[8] Mason (1974). Average lengths computed from Table 13, length frequencies by age (2 mm groupings). Peak migration date is for a one-week period, commencing on the day given.

Table 50. Length and migration timing of outmigrant coho salmon in Pacific Northwest streams.

Stream (state)		$\bar{L}$	Migration timing Median	 Peak	Operation period
Waddell Creek (CA)[1]					Year round
	1934	113.4	4/22	4/22	
	1935	112.9	5/06	5/06	
	1936	109.1	5/06	5/06	
	1937	116.2	5/06	5/06	
	1938	114.8	5/20	5/20	
	1939	115.7	4/22	4/22	
	1940	111.9	5/20	5/20	
	1941	109.5	5/06	5/06	
	1942	103.4	5/13	5/13	
	Mean	111.9	5/07	5/07	
Flynn Creek (OR)[2]					Year round
	1964	86.0			
	1965	83.3			
	1966	78.4			
	1967	83.2			
	1968	87.9			
	1969	83.9			
	Mean	83.8			
Big Beef Creek (WA)					
	1978[3]	105.7		5/07	4/21-6/02
	1979[3]	104.7		5/08	4/02-6/16
	1980[3]	105.7		5/06	3/08-6/19
	Mean	105.4		5/07	
Minter Creek (WA)[4]					
	1940			∿4/30	Year round
	1942			∿5/16	
	1945			∿5/05	
	1946			∿5/20	
	1948			∿5/04	
	1950			∿5/06	
	1951			∿5/06	
	1953			∿5/04	
	1954			∿5/08	
	1955			∿5/12	
	Mean			5/08	
Kelsey Creek					
	1981	109.1	4/18	4/27	Year round
Tenderfoot Creek (B.C.)[5]					
	1973	73.0	3/10	5/14	4/18-3/23
	1974	97.0	5/15	5/28	4/11-6/04
	1975	84.0	5/24	5/12	4/09-6/15
	Mean	84.7	5/16	5/18	

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[1] Shapolov and Taft 1954. Migration timing dates were calculated from Table 15, and are for a one-week period (e.g. 3/11 includes fish captured up to 3/18).

[2] Au 1972. Average lengths calculated from data presented in Table A-9, weighted by catches in Table A-4.

[3] Seiber et al. 1981.

[4] Salo and Bayliff 1958. Migration timing dates from Figs. 19-23.

[5] Armstrong and Argue 1977.

would rear an additional year in Lake Washington.

This hypothesis may be tested by an analysis of the scales of adult cutthroat, assuming year class strength was relatively constant and that differential mortality did not occur. A review of 81 cutthroat spawner scales, collected at the net site, revealed that only 7 percent of the fish had spent an entire year in the lake before entering the ocean, and 2 percent had failed to leave the lake (Appendix Table 31). Fish which had spent one year in the stream comprised 58 percent of the sample, and all remaining fish had reared in the stream for 2 years.

The date of peak outmigration for cutthroat and coho in Kelsey Creek (Tables 49 and 50) appears to be slightly earlier than in other streams for which data are available. Moring and Lantz (1975) noted that cutthroat in the Needle Branch migrated from that stream at an earlier date after logging had removed riparian vegetation. They hypothesized that the shift in migration timing was in response to alteration of the temperature regime. Water temperatures in Kelsey Creek are consistently greater than in Bear Creek, but several more years of study would be needed to test the applicability of this hypothesis to Kelsey Creek.

It is unfortunate that the net was not fished through at least one of the major floods of 1980-1981. The maximum discharge at which migration was monitored, 4.27 m^3/sec, is less than 30 percent of the peak recorded at Kelsey Creek in the 1979 and 1980 water years. Up to a discharge of 4.27 m^3/sec, it is evident that species with relatively poor swimming ability, rather than salmonids, are being swept from the system. Storms on both August 17 and November 26 increased markedly the catch of the large-scale sucker-bluegills, and ammocoetes.

Fish in Kelsey Creek showed no tendency to actively migrate from the stream when the potential for first flush phenomena was present. Catches on October 24 and July 28 were no different than those on the previous two sampling dates. Catches of large-scale sucker and other weak swimming species did increase on August 26, but as noted above, this was undoubtedly related to the rise in discharge rather than pollutants.

8.5.2 Instream Population Dynamics

Most ecologists would agree that stream environments are highly dynamic and, in some cases, inherently unstable as they are subject to periodic disruptions of a natural tendency toward equilibrium. The trend to increase stability through adjustments of channel morphology and hydraulics (Langbein and Leopold 1966) may be inhibited by continuing environmental disturbances. Similarly, the structural and functional characteristics of the aquatic community, which also tend towards homeostasis, are often disrupted by abiotic phenomena. Vannote et al. (1980) defined stream ecosystem stability as "a tendency for reduced fluctuations in energy flow, while community structure and function are maintained, in the face of environmental variations." Implicit in this concept of ecosystem stability is the capacity of the community to resist environmental disturbances or, in view of the dynamic quality of natural equilibria, the ability of the community to return to its former state when stress (or change) is relieved (Porcella and Sorensen 1980). Assuming that stream communities are adapted to the most probable set of conditions encountered in the stream, the maintenance of organization and the efficiency of energy utilization in the aquatic community is largely determined

The destabilizing influences of urbanization as they affect hydrologic processes and sediment dynamics have been detailed in the preceding pages. The effects of increased runoff intensity and volume during storm events act to reduce the equilibrium capability of the urban stream. Variations in sediment loading resulting from construction activities, nutrient enrichment, the introduction of toxic substances, channelization, and the removal of shade and cover-producing vegetation along the stream are additional perturbations which enhance instability.

Whereas factors contributing to the instability of the physical system are relatively easy to identify, their combined effect on the associated stream community is difficult to measure. For example, it cannot be assumed _a priori_ that the widely fluctuating environment of an urban stream is inimical to a multiple and diverse fauna, although several studies have shown that fluvial systems impacted by urban runoff tend to be species poor (Slobodkin and Sanders 1969, Tramer and Rogers 1973). Vannote et al. (1980) point out, however, that a high degree of physical variation, provided that lethal conditions are not encompassed, may provide optimum conditions for a large number of species. Complexity is not necessarily an attribute of stable ecosystems (May 1973) and, while the diversity and functional complexity of an altered stream may be depressed relative to its former state, the measurement of these parameters is not an acceptable index of ecosystem stability.

A suitable starting point in the analysis of ecosystem stability would be to study the action of a physical factor, or the joint action of many factors, by observing the reaction of individual species (or populations) within the stream community. This makes sense intuitively when one considers that environmental stress can be defined only in regard to a particular species;

in other words, a habitat stable for one species may be unstable for another (Krebs 1972). The reaction of a species to an array of environmental variables can be examined quantitatively through its population dynamics, or, more precisely, by the basic processes of birth, death, immigration, emigration, and growth. According to LeCren (1969), production, because it incorporates statistics on age, growth, density, mortality, and biomass, is the best aggregate measure of the ecological performance of a fish species in an aquatic ecosystem. A high incidence of perturbation can result in irregular fluctuations in population numbers and production from one year to the next. However, if a more or less complicated cycle of annual production is observed in a species with several age groups, then endogenous (i.e., intraspecific) regulation may be maintaining an imbalance caused by abiotic fluctuations sometime in the past (Bohlin 1977).

Loss (L) and replacement (R) rates are a direct measure of the stability, in a spatial sense, of fish populations. The calculated rates for cutthroat trout differed little between Kelsey and Bear Creeks, and indeed were often somewhat less in Kelsey Creek for age 0 fish. Sample sizes for coho salmon on Kelsey Creek were extremely small, but similar conclusions appear to be applicable.

The relative biomass of salmonids is high in Kelsey Creek and low in Bear Creek in comparison with estimates for other streams in the northwest. The combined biomass of coho and cutthroat in Kelsey Creek ranged from 1.8 to 6.5 $g{\cdot}m^{-2}$ during 1979-1981, while Bear Creek salmonid biomass varied between 1.3 and 3.7 $g{\cdot}m^{-2}$ for the same period.

Lowry (1966) estimated that an average of 7.8 $g{\cdot}m^{-2}$ of cutthroat and coho biomass was present in three Oregon streams during a 20-month study period, indicating that the salmonid biomass for Kelsey Creek is not exceptionally high.

Using Edie's (1975) data on teleost standing crops in eight tributary streams of the Clearwater River in western Washington, a range of 0.0 to 6.15 $g{\cdot}m^{-2}$ in salmonid biomass during the summers of 1973 and 1974 was calculated. The mean salmonid biomass of the eight streams, including one which had been stocked with coho fry, was 2.31 $g{\cdot}m^{-2}$ for both years. This value represents less than a third of the average biomass of the entire assemblage of fishes sampled. Nonsalmonids were similarly abundant in Bear Creek, and the presence of these species may have the effect of reducing salmonid production below a level which would be attained in their absence.

Prior research on the characteristics of salmonid populations in urban streams has been limited, thus the generality of the results obtained in Kelsey Creek is unknown. However, many of the physical processes impacted by urban development, including those related to sediment production, hydrology, and solar irradiation, are affected similarly by forest harvesting, for which an extensive body of literature does exist. This suggests that the dynamics of salmonid populations in logged watersheds could serve as a useful analog for urban stream studies.

The relative susceptibility of coho salmon and cutthroat trout to environmental modification has not been answered by the studies of logging-fish interactions conducted to date. The research by Edie (1975) cited above concurs with the Kelsey Creek data in that the biomass of coho salmon supported in streams impacted by logging was less than that reported for most other areas, while trout (steelhead and cutthroat) biomass appeared normal. In contrast, the cutthroat population in the Needle Branch, Oregon, was depressed for eight years following the logging of the watershed, while the coho salmon population was unaffected (Moring and Lantz 1975).

Numerous studies indicate that logging need not be harmful, and in many cases may be beneficial to salmonid populations, if measures are taken to ensure watershed protection. Much as was observed in Kelsey and Bear Creeks, Murphy and Hall (1981) found that the biomass of cutthroat trout at sites on six Oregon streams located in logged basins averaged nearly twice that of the unlogged control sites. The biomass of salmonids in two California streams (Bummer Lake Creek and the South Fork Yager Creek) also increased after a controlled logging program had been conducted (Burns 1972). Chapman and Knudsen (1980) surveyed numerous streams in western Washington and found that in some instances habitat alteration increased the biomass of salmonids supported by the stream. Each of the authors cited above stated that the most beneficial aspect of habitat alteration was the increase in solar irradiation reaching the stream, a result of the removal of a significant portion of the overhanging canopy.

The riparian vegetation along the middle and lower reaches of Kelsey Creek constitutes only a fraction of its former growth. The removal of streamside cover has not resulted in excessive water temperatures and appears to have indirectly benefitted the trout population in the urban stream. An increase in the amount of insolation, higher nutrient concentrations, and larger substrate size are important physical variables affecting primary production and, by energy transfer through the food web, salmonid production in Kelsey Creek. Also important are the opportunistic, if somewhat functionally limited, benthic organisms found in Kelsey Creek. The benthos consists primarily of suspension and deposit-feeding oligochaetes, chironomids, and amphipods, all of which were represented in the diet of juvenile cutthroat and coho. These taxa subsist on small particulate organic matter (e.g., algal detritus) and are apparently well adapted to the unstable environment of Kelsey Creek (Pedersen 1981).

Bear Creek, which is heavily canopied along most of its course, fits the general description of a light-limited stream. Maximal fish growth coincides with the large energy input to the stream from leaf fall during the autumn months. This contrasts with Kelsey Creek, where fish growth is stimulated during the spring and early summer when periphytic and, presumably, benthic production is greatest. The composition of the invertebrate fauna in Bear Creek is much more diverse than in Kelsey, including high percentages of leaf shredders and predators typical of salmonid streams, although the overall abundance of benthos is similar in the two streams (Pedersen 1981). However, the assemblage of organisms in the urban stream, particularly the chironomids, are characterized by comparatively short generation times and, thus, benthic production may be expected to be higher than in Bear Creek.

Regardless of the relative production of benthic invertebrates in each stream, it is evident that salmonids grow more rapidly in Kelsey Creek than in Bear Creek. The size of an age I outmigrant cutthroat trout from Kelsey Creek is near the length of age II outmigrants from other systems.

The combined annual production of cutthroat and coho in Bear and Kelsey Creeks can be compared to production estimates for other salmonid streams (Table 51) which have been summarized by LeCren (1969). The highest annual production recorded to date for a stream-dwelling salmonid is 18.8 $g \cdot m^{-2}$, estimated for a brown trout population in a small Danish stream (Mortensen 1977). This is by no means the highest production estimate for an individual species in lotic systems: Naiman (1976) reported that the herbivorous desert

Table 51. The annual production of salmonids for selected streams of the world.

Author	Stream	Species	Annual production $g\ m^{-2}\ yr^{-1}$
Mortensen (1977)	Denmark	*S. trutta*	18.8
Hopkins (1971)	New Zealand	*S. trutta* *Phylipnodon breviceps*	14.5
Hanson and Waters (1974)	Minnesota	*Salvelinus fontinalis*	14.4
Lowry (1967)	Oregon	*Salmo clarki* *Oncorhynchus kisutch*	13.0
LeCren (1969)	England	*S. trutta*	3.0-12.0
Hunt (1974)	Wisconsin	*S. fontinalis*	11.7
O'Connor and Power (1975)	Ontario	*S. fontinalis*	1.4-6.6
Power (1973)	Norway	*Salmo salar* *S. trutta* *Salvelinus alpinus*	0.1-6.14
June (1981)	Washington	*S. clarki*	1.97

pupfish, Cyprinodons nevadensis, produced 155.4 $g{\cdot}m^{-2}$ of tissue annually as the only fish species in a warm desert stream.

In a study of several infertile, high-latitude streams in northern Norway, the annual production of coexisting and separate populations of Atlantic salmon (Salmo salar), brown trout, and char (Salvelinus alpinus), ranged from 0.10 to 6.14 $g{\cdot}m^{-2}$, including the lowest production estimates yet made for self-sustaining populations of salmonids. In three Pacific Northwest streams Lowry (1966) calculated a combined annual production for cutthroat and coho populations of about 13.0 $g{\cdot}m^{-2}$, comparable to estimates of 11.7 and 14.4 $g{\cdot}m^{-2}{\cdot}yr^{-1}$ for brook trout (Salvelinus fontinalis) in two midwestern streams (Hunt 1974, Hanson and Waters 1974). In a recent study of sympatric populations of resident and sea-run cutthroat trout in a coastal stream in Washington, June (1981) estimated the total annual production to be 1.97 $g{\cdot}m^{-2}$.

Some indication of the variability in production between streams, part of which is undoubtedly due to methodological differences among researchers, may be inferred from the foregoing synopsis. Given the range of previous estimates, the production of salmonids in Kelsey Creek, while greater than in Bear Creek, is only moderate in magnitude. Salmonid production in Bear Creek is low by the same standards, the direct consequence of a depressed standing crop.

The factors which cause, and the effects of the respiratory anomalies prevalent on fish sampled from Kelsey Creek and the West Tributary will require additional research. Although spurious correlations are possible, it is interesting to note that incidence rates generally increased with the degree of urban development within a watershed, while the density of salmonids decreased (Table 52).

Table 52. Land use, salmonid density, and the percentage of age 0 cutthroat trout afflicted with respiratory anomalies in the Kelsey Creek watershed.

Stream	Urban index[1]	Salmonid density (No. m^{-2})	% age 0 afflicted
Valley Creek	2[2]	2.93	6
Richards Creek	21	0.87	5
Kelsey Creek	24	0.60	68
West Tributary	31	0.11	50

[1] Percentage of land within the watershed occupied by institutions, industry, commerce, and freeways (data from Richey 1982).

[2] Includes only the portion of the watershed above the sampling site.

8.5.3 Ecological Success and Stability

Because of their position near the top of the aquatic food chain, fish are an important component in the energy relations of stream communities. Fish production is a direct consequence of changes in growth and biomass over time, variables which reflect the influence of environmental conditions, whether natural or induced by man, on the density, mortality, growth, and reproduction of a population. As such, the analysis of production dynamics is a useful means of evaluating the relative ecological success of two or more species sharing the same environment. One conclusion from this study is that cutthroat trout, possibly because they are more able to cope with the stress caused by urbanization, are the most successful salmonid species in Kelsey Creek. From a consideration of its stable growth and mortality, high biomass and production values, coho salmon are the dominant species in Bear Creek.

Estimates of production and its component parameters over short time intervals, however, give little indication of the long-term stability of a population, which was defined at the beginning of this chapter as the ability of a population (or community) to regain its former state following environmental perturbation. The concept of stability has been used to refer to a number of ecological attributes but in this study it was chosen to mean the maintenance of organizational integrity. Organization is maintained within a long-lived population by a stable age distribution, characterized by a regular survivorship curve (Chadwick 1976) which ensures reproduction adequate to maintain the population structure. The number of fish in each age group declines with increasing age in the Kelsey Creek cutthroat population, implying stability, but the Bear Creek trout population does not follow this pattern and should be considered unstable. We would expect the younger fish in a stable population to achieve greater production since surplus energy available for growth normally decreases with age. Again, the trout population exhibited these trends in Kelsey Creek but not in Bear Creek. A final attribute of a stable population would be small variations in annual production, implying a long-term efficiency in energy utilization. Hunt (1974) found that annual production in a brook trout population in a Wisconsin stream varied by less than 20 percent over a five-year period. The annual production in Kelsey Creek computed for 1979 differed from that of 1980 by only 12 percent, whereas the annual production in Bear Creek increased by over 50 percent in 1980.

9.0 SUMMARY

A three-year study was initiated in January 1979 to assess the effects of urban development upon the aquatic biota of Kelsey Creek, Bellevue, Washington. This project was a joint effort of the Department of Civil Engineering and the Fisheries Research Institute of the University of Washington. Detailed within this report are the results which pertain to the fisheries resources of Kelsey Creek.

Urban development has proceeded rapidly in the Kelsey Creek drainage basin since the early 1950's, and currently only 22 percent of the land within the watershed is undeveloped. Concommitant with this development, peak annual discharges in Kelsey Creek have increased significantly, with floods which previously occurred with a recurrence interval of 10 years now occurring at least every other year. Peak daily water temperatures within the urban stream were found to be greater than those of Bear Creek, the control stream, and diurnal fluctuation was accentuated.

Streambed substrate quality can be an important determinant of the survival of salmonid embryos. The percentage of fine sediment within the streambed of both Kelsey and Bear Creek varied on an annual cycle in response to hydrologic events. Substrate composition was significantly coarser during the winter months, but varied locally in response to stream gradient. The percentage of fine sediment in substrate samples from each stream differed little, both being near the upper limit for undisturbed watersheds previously studied. Intragravel dissolved oxygen concentrations were significantly correlated with the percentage of fine sediment within the sample area.

Two years of sampling indicated that the intragravel dissolved oxygen concentrations of Kelsey Creek were characteristic of disturbed watersheds, and below those conducive to salmonid embryo survival. Instream embryo bioassays verified the relationship between embryo survival and intragravel dissolved oxygen concentrations. The survival of coho salmon embryos was significantly better in Bear Creek than in Kelsey Creek, but no difference was found between the streams in the survival of rainbow trout embryos. Streamside bioassays revealed that the surface water of Kelsey Creek has not been degraded sufficiently to reduce the survival of salmonid embryos.

Modification of the hydrologic regime of Kelsey Creek has resulted in a channel disequilibrium, as evidenced by the frequency and depth of streambed scour. Scour in the urban stream was significantly greater than in the control stream during periods of storm runoff. However, a predictive model which was developed suggests that the mortality of salmonid embryos due to scour is minimal, averaging 13 and 14 percent for coho salmon and cutthroat trout, respectively, in the years 1978 to 1981.

The biomass of juvenile salmonids supported by Kelsey Creek varied from 1.8 to 6.5 $g \cdot m^{-2}$, but was generally greater than that in the control stream. The distribution of the biomass between the various salmonid species-age classes differed between the streams. The majority of salmonid biomass in Kelsey Creek consisted of age 0 and I cutthroat trout, while in Bear Creek, other age classes of cutthroat trout, as well as coho salmon, made significant contributions. Bear Creek also supported a greater biomass of nonsalmonids than did the urban stream.

Growth of salmonids in Kelsey Creek, particularly in the period from emergence to midsummer, was greater than that recorded in Bear Creek. The greater biomass and growth rates of salmonids in Kelsey Creek resulted in an annual net production of up to three times greater than that for the control stream. The total annual salmonid production for Kelsey Creek for the years following July 23, 1979 and 1980, was estimated as 6.6 and 7.4 $g \cdot m^{-2}$, respectively.

The percentage of urban development within the four constituent watersheds of the Kelsey Creek basin influenced the characteristics of the inhabiting salmonid populations. Coho salmon were absent from streams draining heavily developed areas, and the incidence of cutthroat trout with respiratory anomalies increased. The density of salmonids was greatest where development was minimal.

The unstable environment characteristic of Kelsey Creek did not appear to result in the voluntary or forced displacement of the salmonids inhabiting the stream. Loss and replacement rates, two indices to the spatial stability of fish populations, did not differ markedly between the control and study streams. Outmigrant netting at the mouth of Kelsey Creek revealed no tendency for fish to migrate from the stream when the potential for first flush phenomena was present, nor did floods of moderate magnitude appear to result in forced displacement.

The smolt production for Kelsey Creek is somewhat lower than for other streams of comparable surface area, but this may be due to the abundance of cutthroat trout relative to coho salmon. The average length of cutthroat trout smolts from the urban stream is close to the average length of smolts one year older from other watersheds. In contrast to the findings of several other studies, the majority of the Kelsey Creek cutthroat trout smolts are age I, perhaps due to the rapid growth which they experience.

In urban streams in which toxicant inputs are minor, it may be expected that development will impact the stream in a manner similar to that of forest harvesting. Researchers in that field have suggested that logging may be beneficial to salmonid populations if measures are taken to ensure watershed protection. Similar conclusions appear applicable to urban watersheds.

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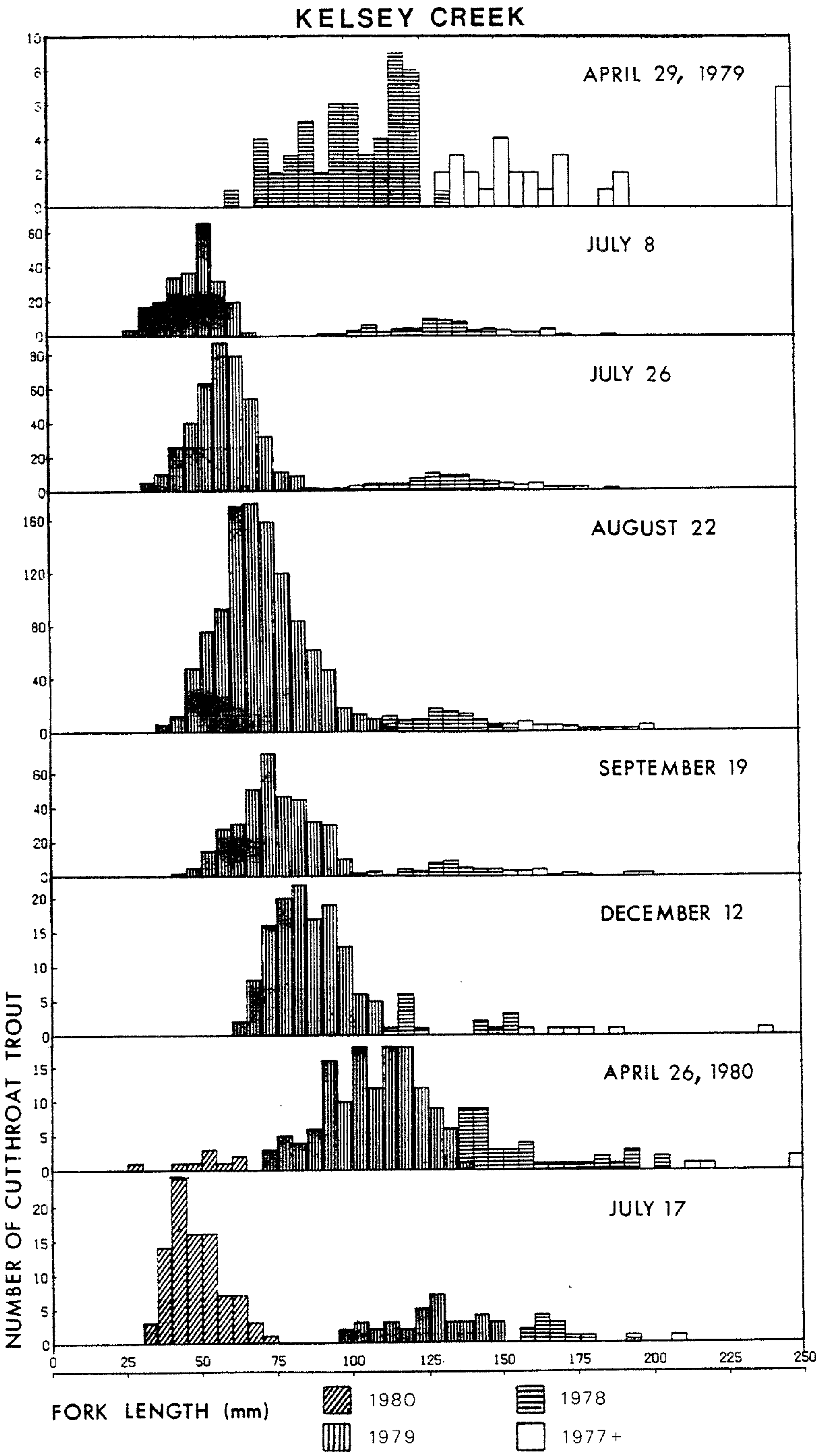
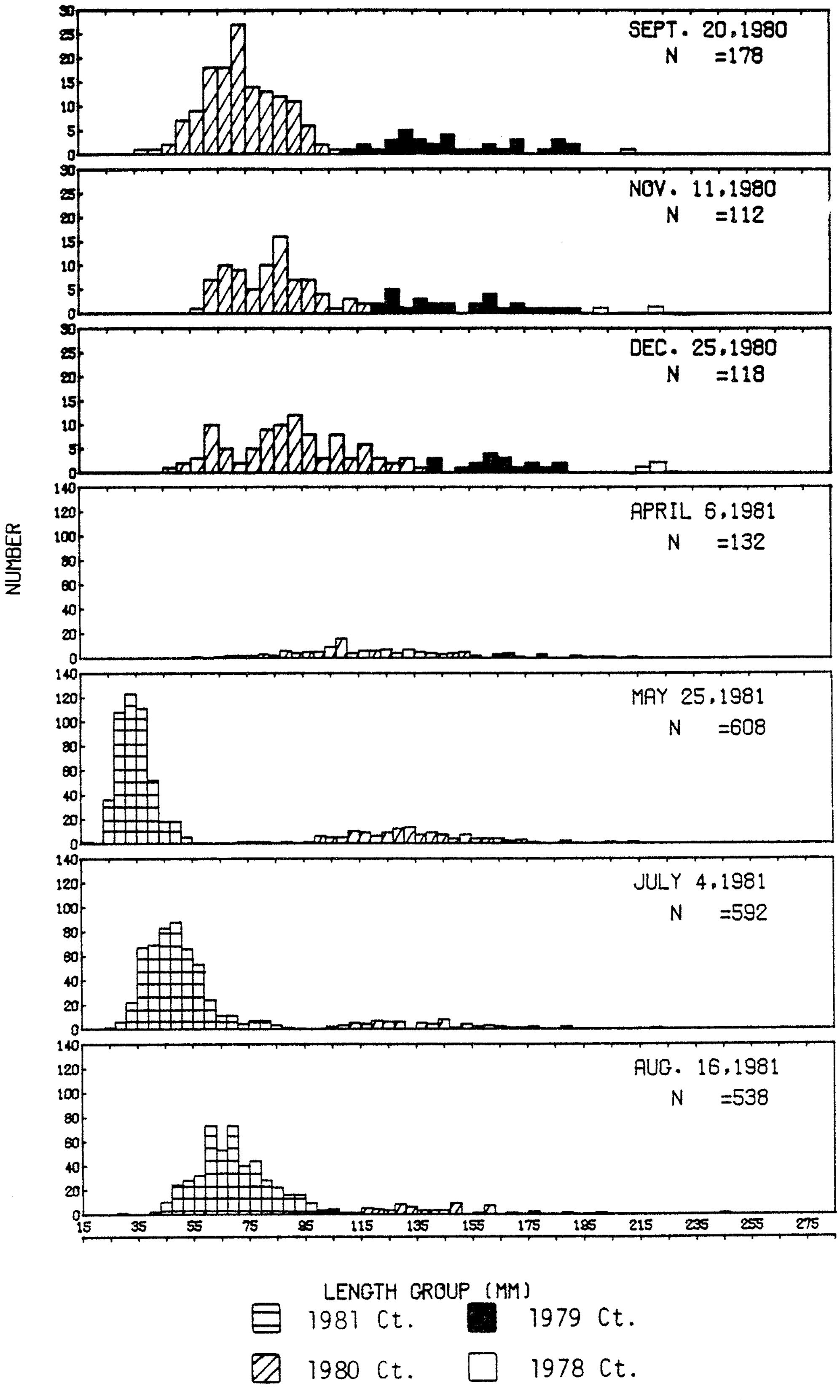
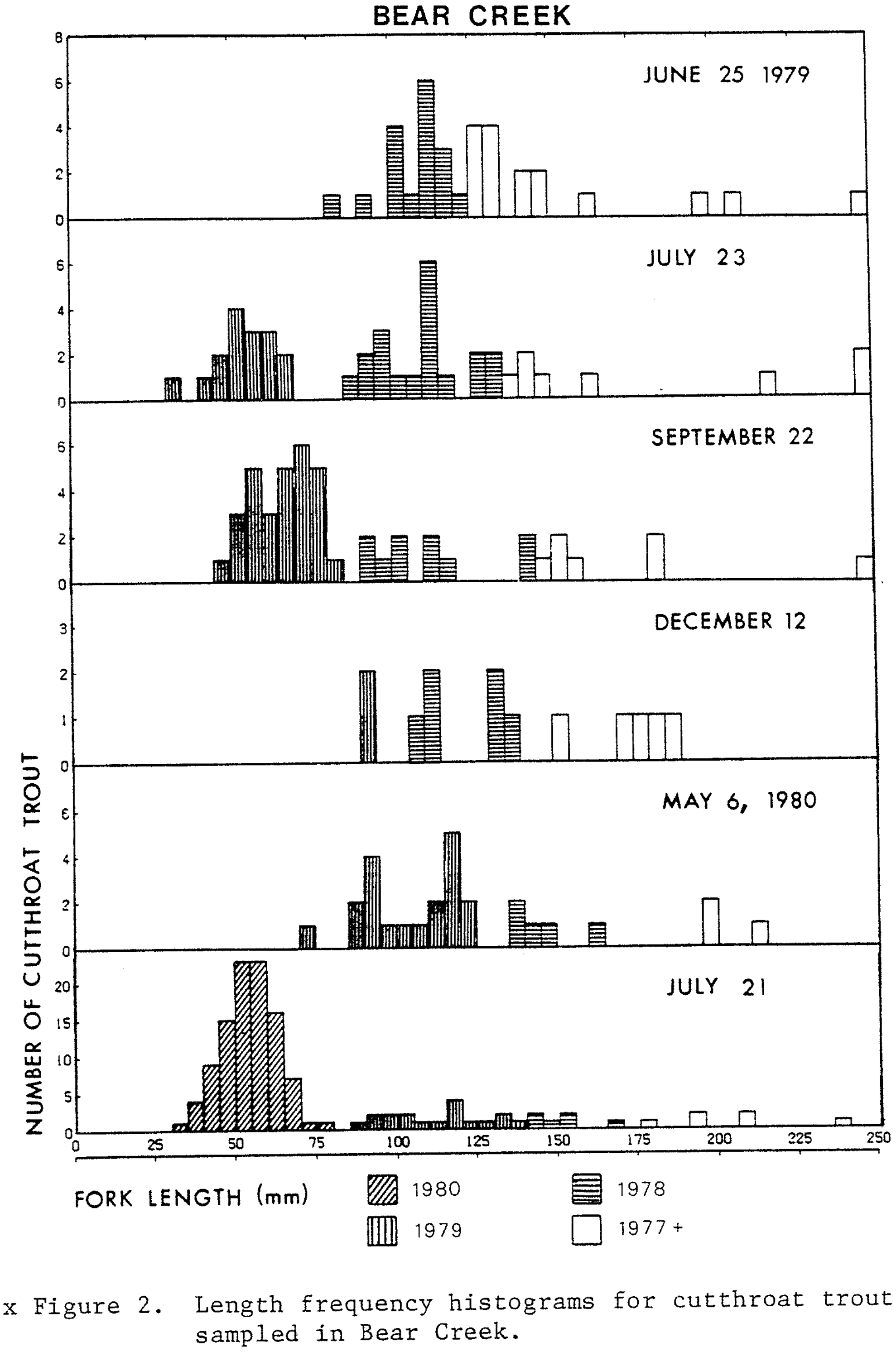
APPENDIX



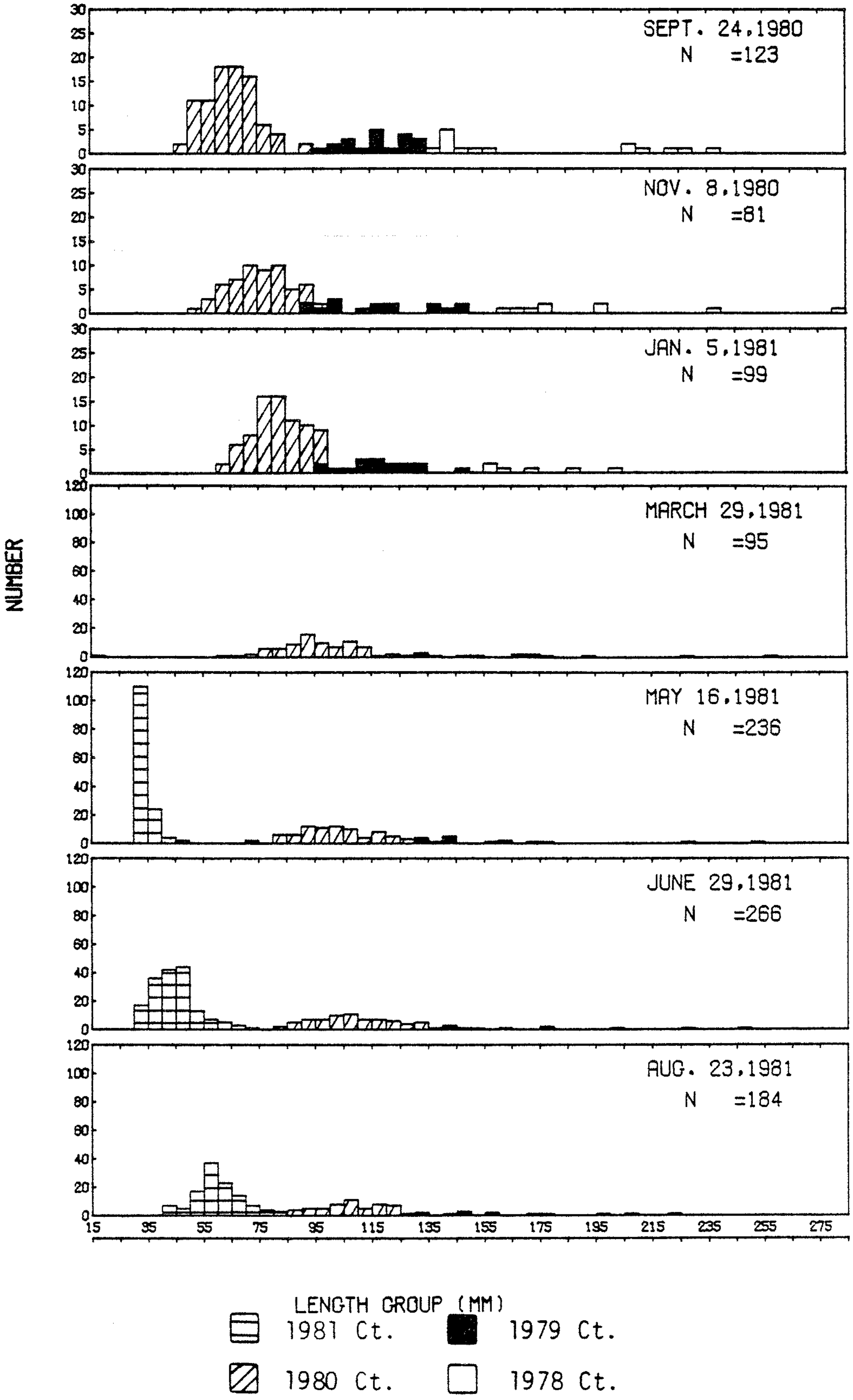
Appendix Figure 1. Length frequency histograms for cutthroat trout sampled in Kelsey Creek.



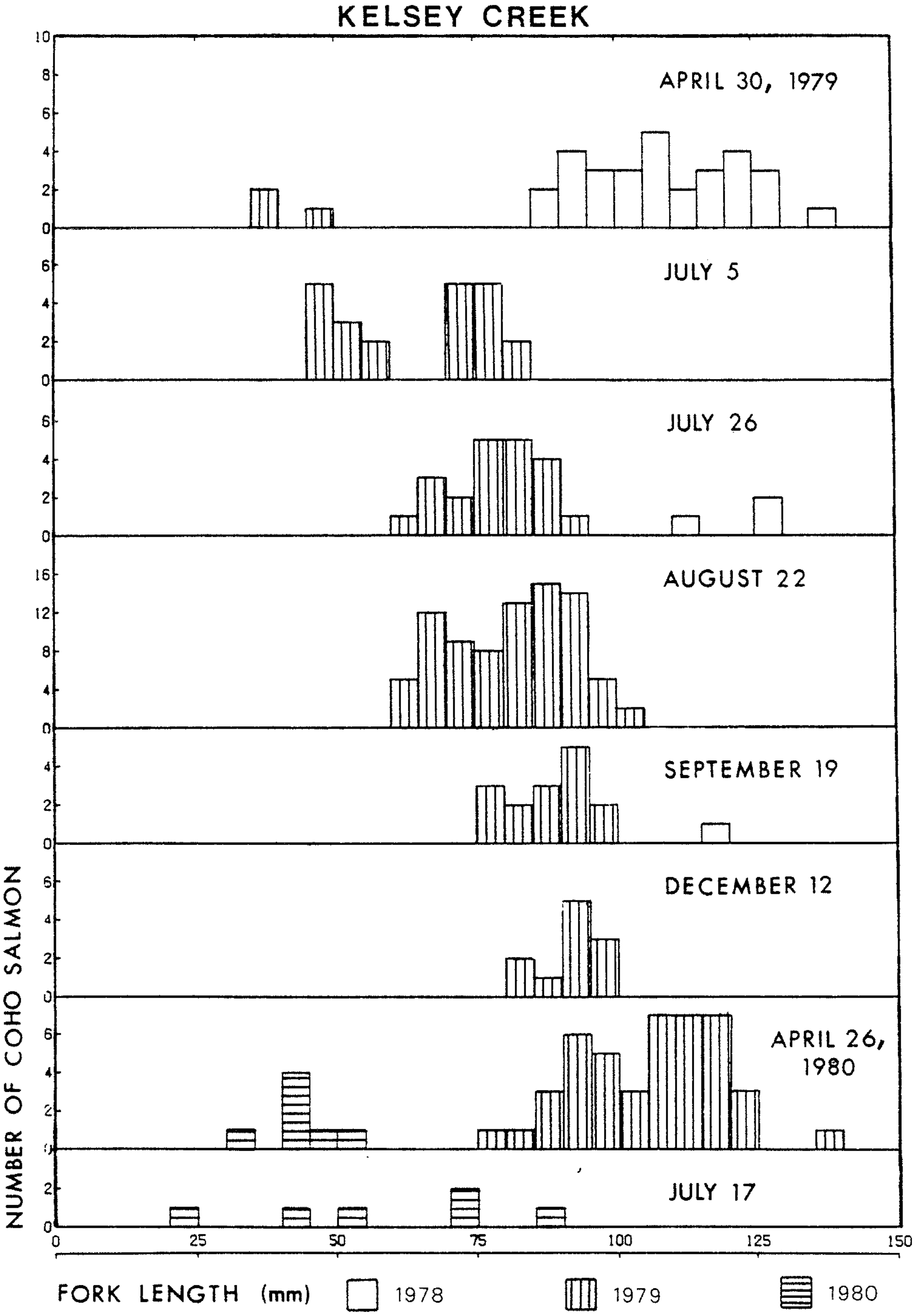
Appendix Figure 1. Length frequency histograms for cutthroat trout sampled in Kelsey Creek (continued).



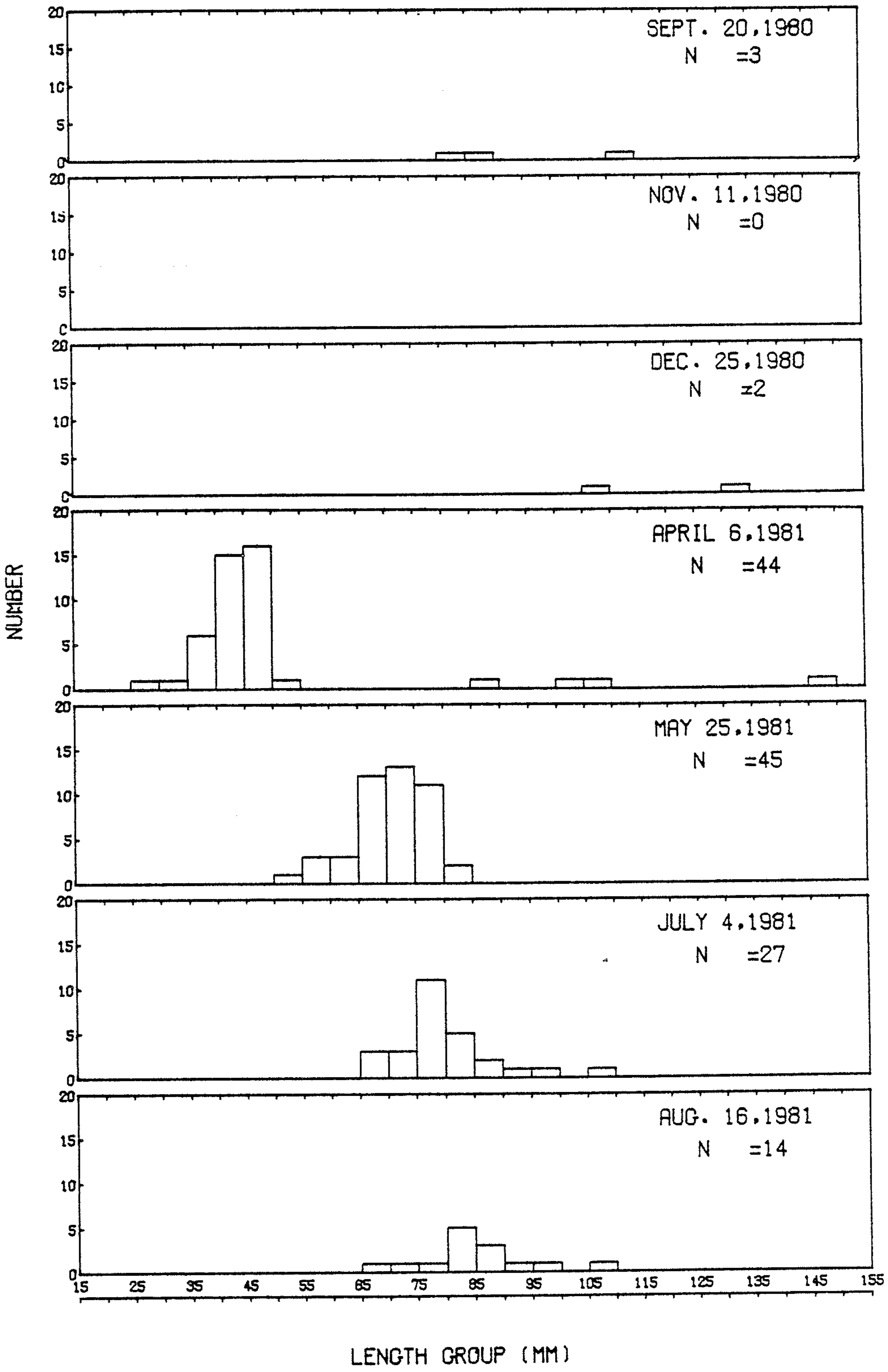
Appendix Figure 2. Length frequency histograms for cutthroat trout sampled in Bear Creek.



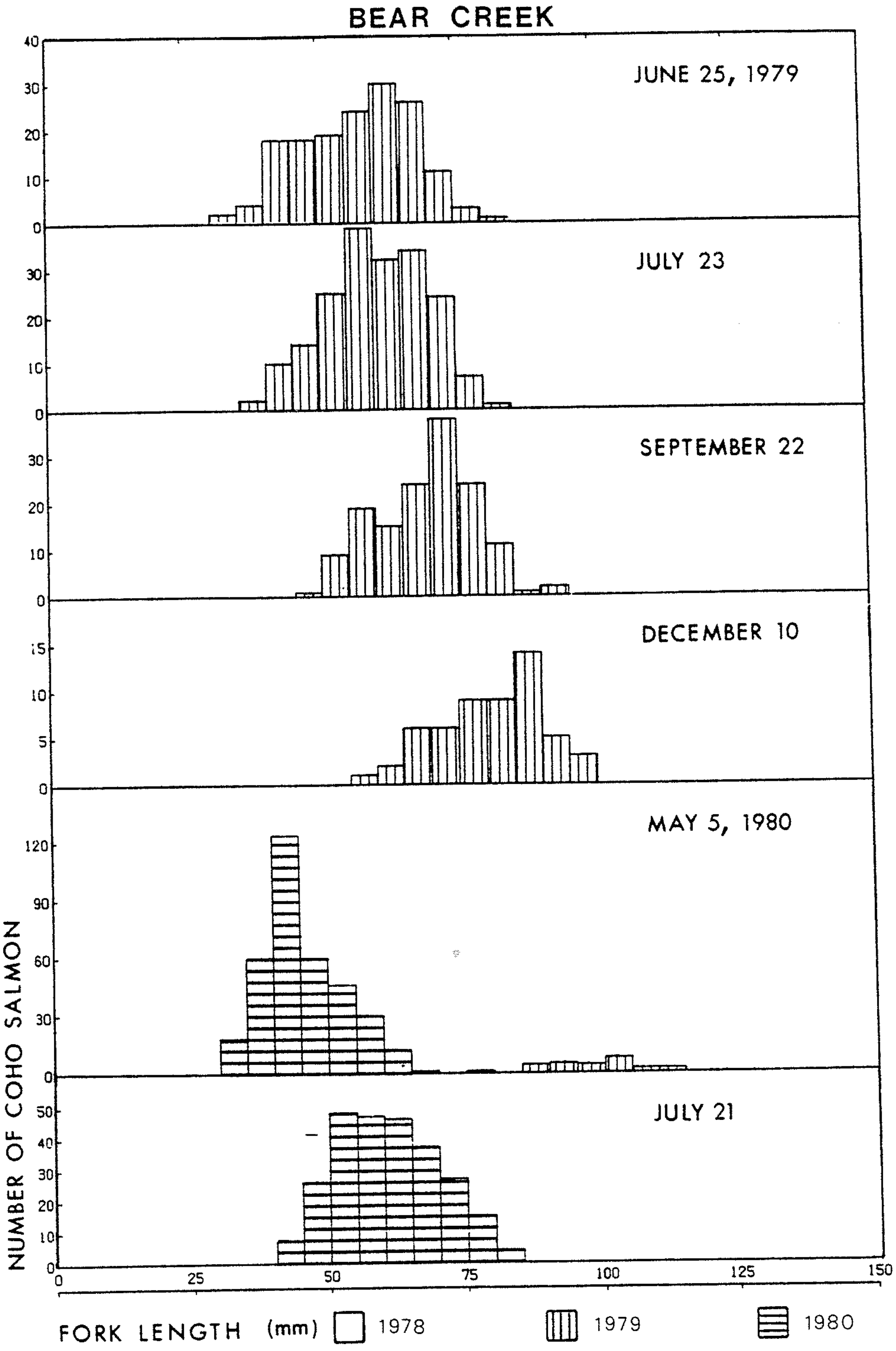
Appendix Figure 2. Length frequency histograms for cutthroat trout sampled in Bear Creek (continued).



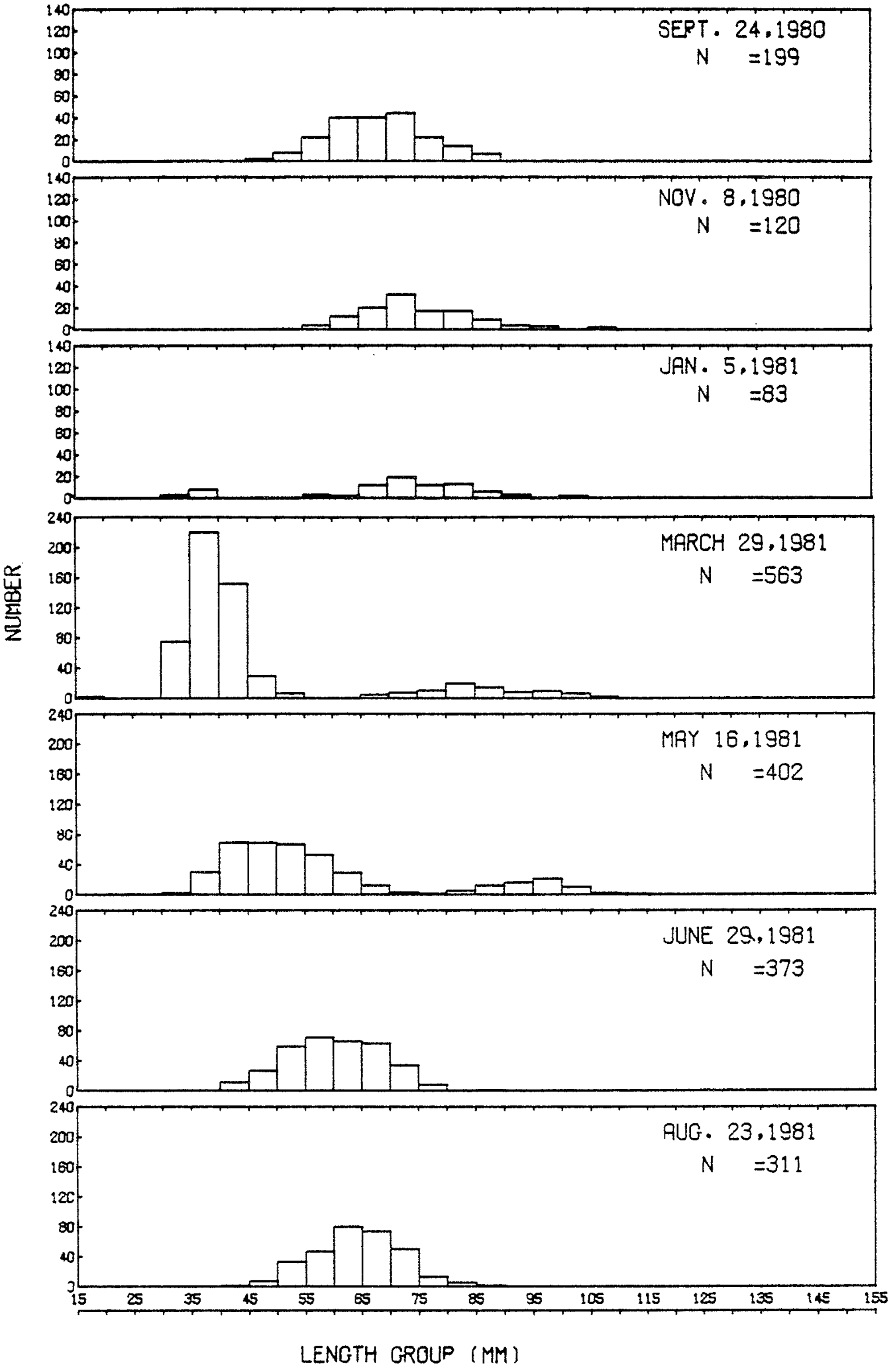
Appendix Figure 3. Length frequency histograms for coho salmon sampled in Kelsey Creek.



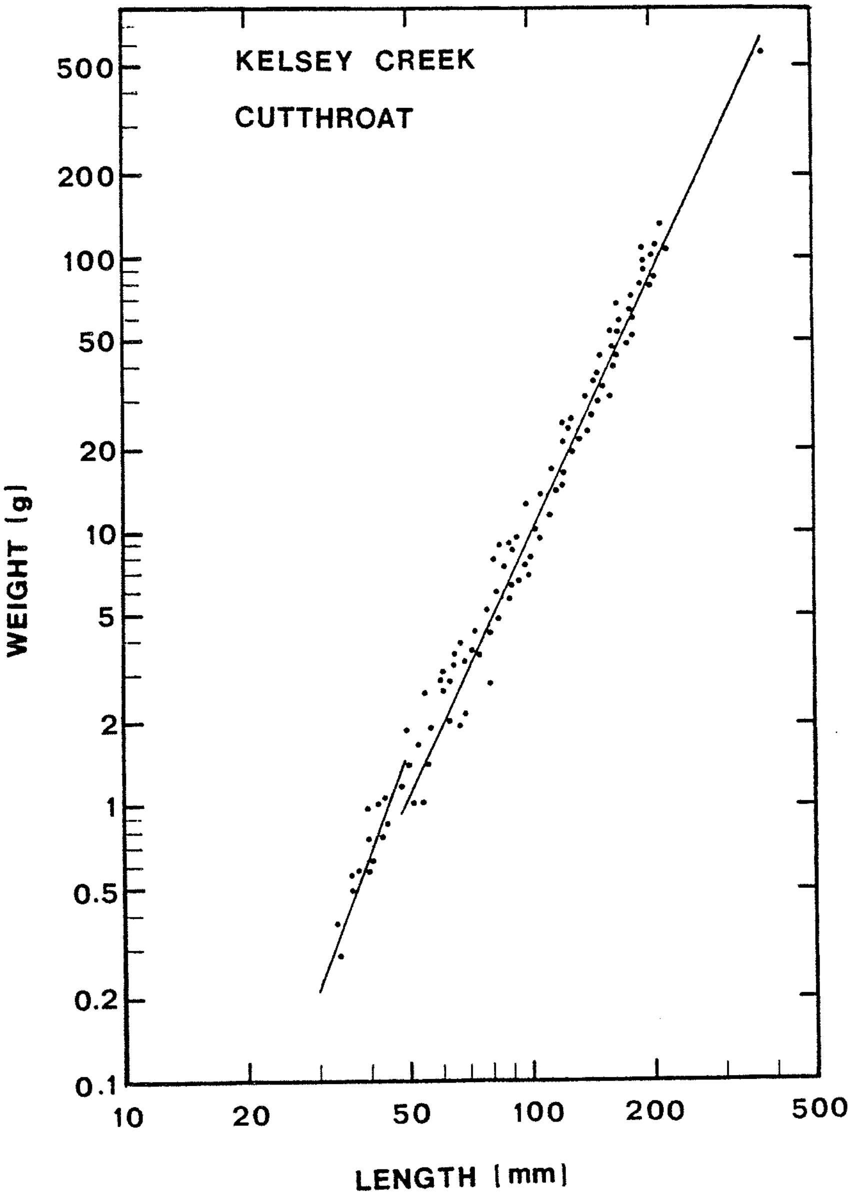
Appendix Figure 3. Length frequency histograms for coho salmon sampled in Kelsey Creek (continued).



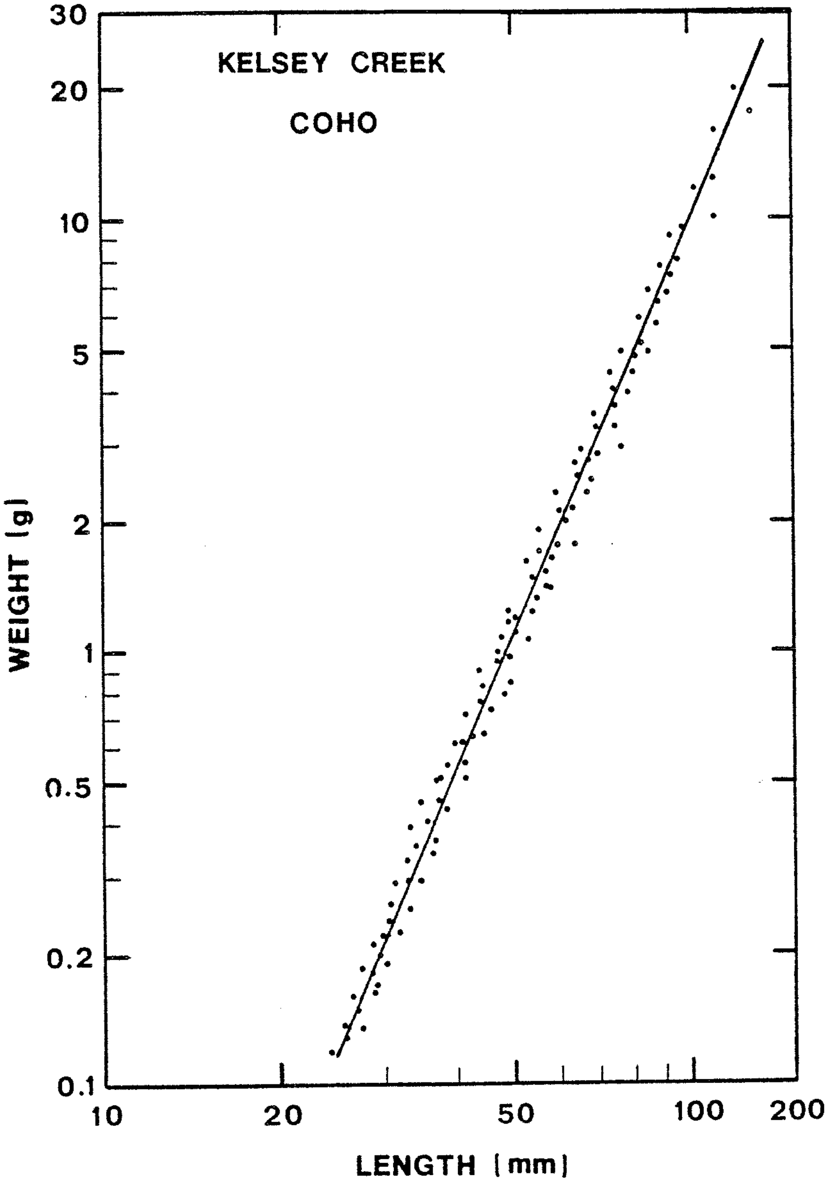
Appendix Figure 4. Length frequency histograms for coho salmon sampled in Bear Creek.



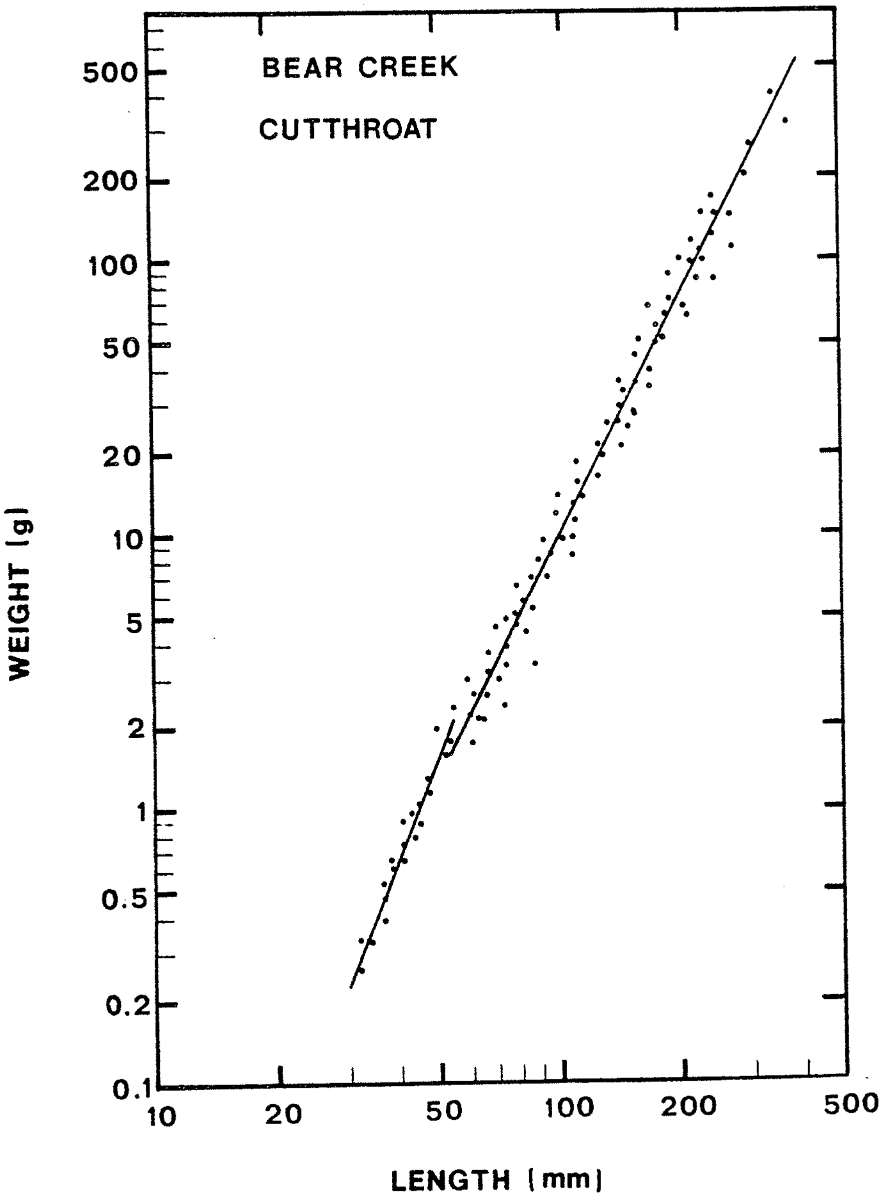
Appendix Figure 4. Length frequency histograms for coho salmon sampled in Bear Creek (continued).



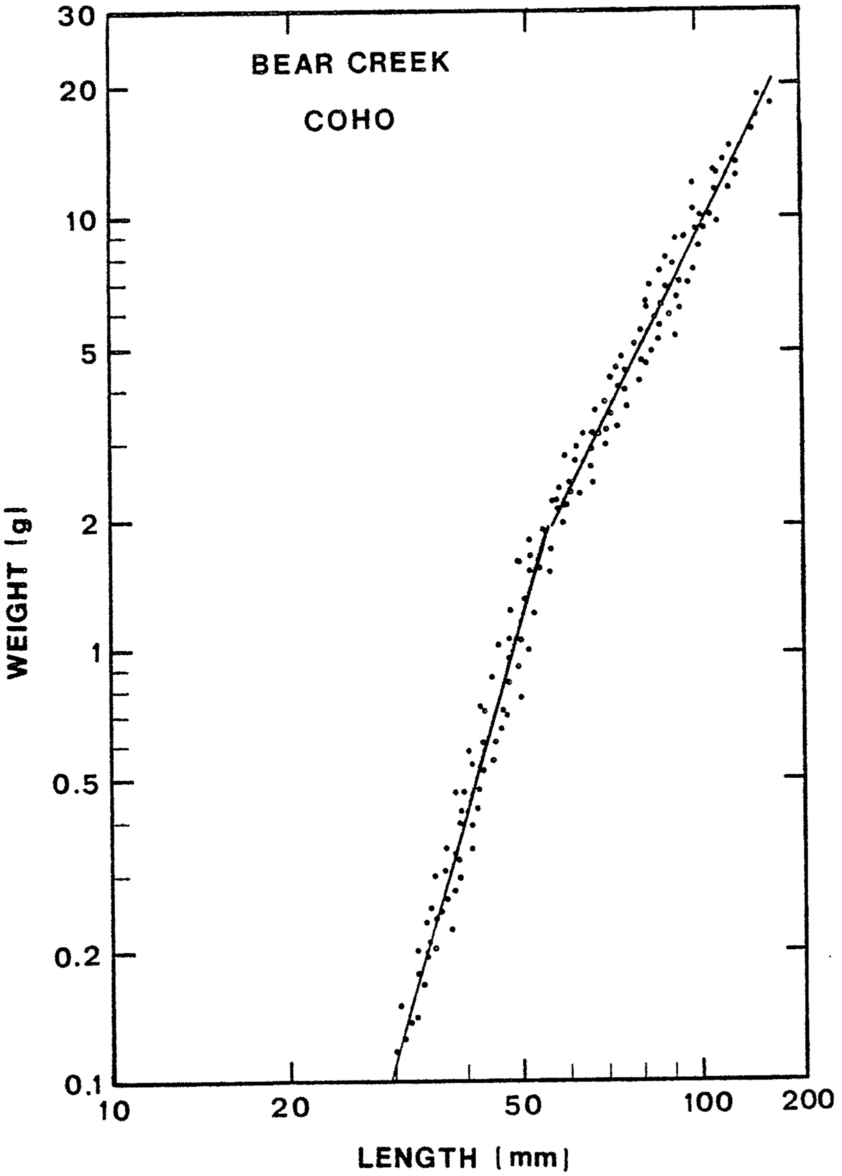
Appendix Figure 5. Length-weight regressions for cutthroat trout captured in Kelsey Creek, April 1979 to July 1980.



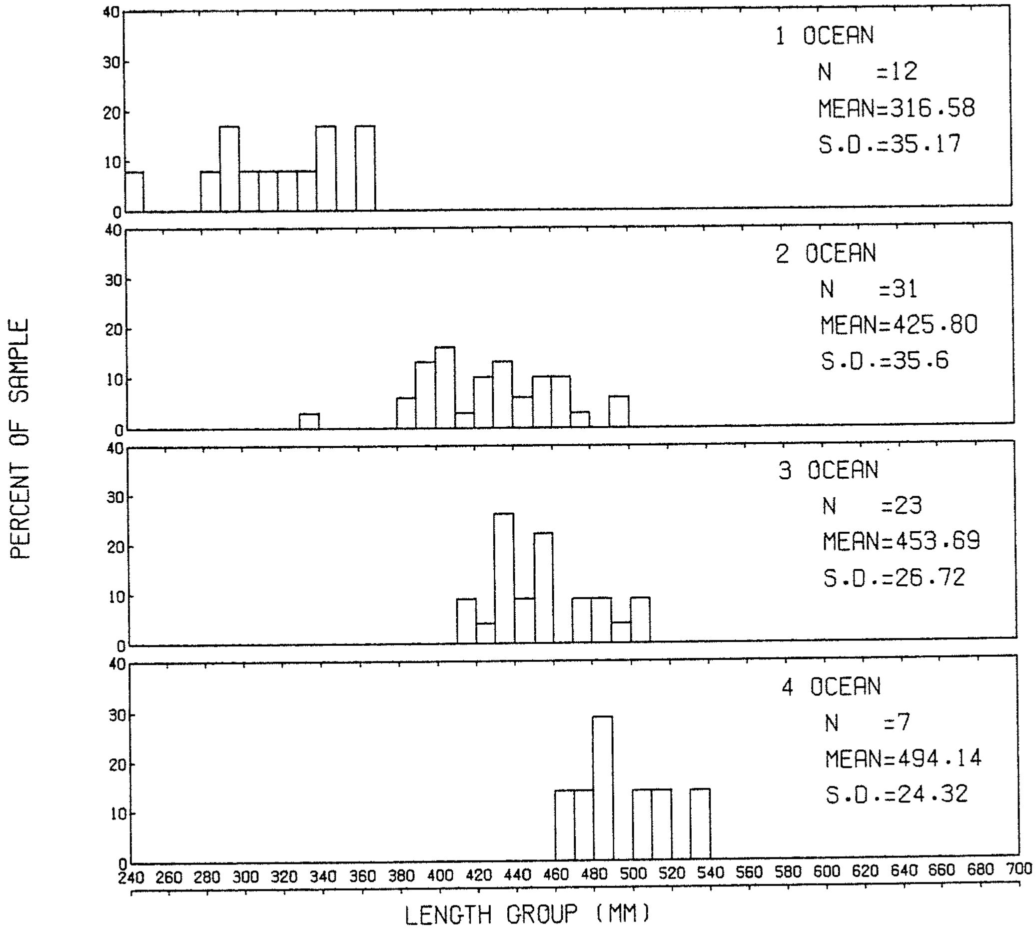
Appendix Figure 6. Length-weight regressions for coho salmon captured in Kelsey Creek, April 1979 to July 1980.



Appendix Figure 7. Length-weight regressions for cutthroat trout captured in Bear Creek, May 1979 to July 1980.



Appendix Figure 8. Length-weight regressions for coho salmon captured in Bear Creek, May 1979 to July 1980.



Appendix Figure 9. Length by ocean age of adult cutthroat trout captured in the downstream net.

Appendix Table 1. Mean concentrations of total ammonia (mg$\cdot$l^{-1} NH$_3$-N) and unionized ammonia (µ$\cdot$l^{-1} NH$_3$) (in parentheses) in interstitial water samples from Kelsey and Bear Creeks, 1979-80.

Date	Temperature	Station							
	°C	KC1	KC2	KC3	KC4	KC5	BC1	BC2	BC3
8/02/79	15	0.04 (0.2)	0.54 (2.9)	0.14 (0.5)	0.27 (1.2)	0.59 (2.5)	0.09 (0.1)	0.02 (0.1)	0.10 (0.7)
9/04/79	15	0.44 (2.9)	0.24 (2.6)	0.37 (3.1)	0.24 (2.0)	0.07 (0.5)	0.22 (1.2)	0.16 (2.1)	0.06 (0.6)
10/05/79	12	0.01 (0.05)	0.54 (2.9)	0.24 (1.3)	0.10 (0.4)	0.61 (3.3)	0.01 (0.03)	0.03 (0.2)	0.10 (0.7)
11/06/79	12	0.14 (0.6)	0.14 (0.6)	0.38 (1.6)	0.23 (0.8)	0.55 (2.3)	0.09 (0.5)	0.17 (0.9)	0.15 (0.4)
12/09/79	9	0.08 (0.3)	0.46 (2.5)	0.14 (0.5)	0.07 (0.2)	0.21 (0.7)	0.06 (0.3)	0.15 (0.8)	0.004 (0.01)
1/27/80	7	0.31 (0.7)	0.12 (0.4)	0.06 (0.07)	0.04 (0.06)	0.05 (0.03)	0.03 (0.04)	0.12 (0.2)	0.09 (0.1)
3/05/80	9	0.03 (0.06)	0.03 (0.04)	0.03 (0.06)	0.02 (0.03)	0.04 (0.07)	0.01 (0.02)	0.01 (0.02)	0.02 (0.05)
4/11/80	11	0.58 (0.5)	0.04 (0.05)	0.03 (0.07)	0.06 (0.11)	0.03 (0.10)	0.02 (0.02)	0.05 (0.03)	0.09 (0.09)
5/05/80	12	0.03 (0.04)	0.20 (0.3)	0.04 (0.1)	0.12 (0.08)	0.08 (0.1)	0.04 (0.08)	0.10 (0.05)	0.06 (0.05)
6/06/80	16	(0.12) (0.3)	0.47 (1.1)	0.16 (0.6)	0.15 (0.4)	0.06 (0.06)	0.03 (0.03)	0.06 (0.09)	0.09 (0.2)

Appendix Table 2. Mean pH levels in interstitial water samples from Kelsey and Bear Creeks, 1979-80.

Date	Station							
	KC1	KC2	KC3	KC4	KC5	BC1	BC2	BC3
8/02/79	7.3	7.3	7.2	7.2	7.2	7.3	7.3	7.4
9/04/79	7.4	7.6	7.5	7.5	7.5	7.3	7.7	7.6
10/05/79	7.5	7.4	7.4	7.3	7.4	7.3	7.5	7.5
11/06/79	7.3	7.3	7.3	7.2	7.3	7.4	7.4	7.1
12/09/79	7.4	7.5	7.3	7.3	7.3	7.4	7.5	7.3
1/27/80	7.2	7.3	7.9	7.0	6.6	7.0	7.0	7.0
3/05/80	7.1	7.0	7.1	7.0	7.0	7.1	7.2	7.1
4/11/80	6.6	6.8	7.1	7.0	7.2	6.8	6.5	6.7
5/05/80	6.9	6.9	6.9	6.7	6.7	6.5	6.6	6.6
6/06/80	6.9	6.9	7.1	7.0	6.6	6.6	6.7	6.9

Appendix Table 3. Mean dissolved oxygen concentrations ($mg \cdot l^{-1}$) in interstitial water samples from Kelsey and Bear Creeks, 1979-80.

Date	Station							
	KC1	KC2	KC3	KC4	KC5	BC1	BC2	BC3
8/02/79	1.42	0.00	1.42	0.18	0.00	1.02	1.32	0.88
9/04/79	3.73	4.66	2.88	4.88	3.02	2.63	4.00	1.10
10/05/79	0.65	1.77	2.33	2.43	0.43	2.30	1.63	2.05
11/06/79	1.70	4.78	5.42	4.73	3.13	4.25	2.40	4.72
12/09/79	4.82	5.75	5.03	7.20	5.30	7.50	3.72	7.95
1/27/80	4.68	4.95	6.12	6.30	3.93	5.78	6.43	7.15
3/05/80	2.73	5.52	4.50	4.70	3.67	4.97	7.40	5.65
4/11/80	6.36	6.20	6.30	4.43	3.23	4.80	3.60	5.90
5/05/80	4.18	3.91	3.90	3.78	2.52	3.83	3.85	4.48
6/06/80	2.50	2.57	4.23	5.77	0.33	3.53	2.23	3.47

Appendix Table 4. Estimated population size, average weight, production, and associated parameters of salmonids at electrofishing sites on Kelsey Creek.

Dates (length)	S.A. (m^2)	Year class-species	Population			Weight			Production	
			N_1	N_2	Z(1/days)	W_1(gms)	W_2(gms)	g(1/days)	p(gms)	p/m^2day
4/29/79-07/08/79 (70)	870.1	1978 Ct	114	57	.0102	12.7	20.3	.0067	791.9	.0130
		1977 Ct	51	14	.0188	41.8	46.5	.0015	286.3	.0047
4/30/79-07/05/79 (66)		1979 Co	4	23	-.0233	0.6	3.0	.0025	5.7	.0001
								Total	1083.9	.0178
7/08/79-07-26/79 (18)	870.1	1979 Ct	306	530	-.0289	1.2	2.0	.0308	242.8	.0155
		1978 Ct	57	67	-.0062	20.3	20.4	.0004	9.4	.0006
		1977 Ct	14	16	-.0059	46.5	44.3	-.0026	0	0
7/05/79-07/26/79		1979 Co	23	36	-.0170	3.0	5.2	.0265	39.2	.0025
								Total	291.4	.0186
7/26/79-08/22/79 (27)	870.1	1979 Ct	530	1116	-.0087	2.0	3.3	.0183	620.2	.0264
		1978 Ct	67	108	-.0013	20.4	20.7	.0005	18.8	.0008
		1977 Ct	16	30	-.0052	44.3	55.7	.0085	173.8	.0074
		1979 Co	36	101	-.0279	5.2	5.3	.0001	2.4	.0001
								Total	815.2	.0347
8/22/79-09/19/79 (28)	870.1	1979 Ct	1116	408	.0178	3.3	3.6	.0027	187.6	.0077
		1978 Ct	108	48	.0125	20.7	20.8	.0003	12.2	.0005
		1977 Ct	30	15	.0068	55.7	47.1	-.0059	0	0
		1979 Co	101	17	.0499	5.3	6.4	.0066	70.6	.0029
								Total	270.4	.0111
9/19/79-12/12/79 (84)	870.1	1979 Ct	408	134	.0087	3.6	5.9	.0058	818.6	.0112
		1978 Ct	48	16	.0086	20.8	25.3	.0023	219.3	.0030
		1977 Ct	15	4	.0103	47.1	54.5	.0017	109.6	.0015
		1979 Co	17	13	-.0030	6.4	6.8	.0008	7.3	.0001
								Total	1154.8	.0158
12/12/79-04/16/80 (136)	870.1	1979 Ct	134	138	.0028	5.9	12.5	.0056	1005.8	.0085
		1978 Ct	16	47	-.0045	25.3	40.7	.0035	307.7	.0026
		1977 Ct	4	5	.0102	54.5	94.0	.0040	201.2	.0017
		1979 Co	13	13	-.0048	6.4	6.8	.0039	82.8	.0007
								Total	1597.5	.0135
4/26/80-07/17/80 (82)	870.1	1980 Ct[1]	0	144	.0299	0.1	1.2	.0276	712.6	.0078
		1979 Ct	138	49	.0187	12.5	18.6	.0048	784.8	.0110
		1978 Ct	47	3	.0167	40.7	49.5	.0024	378.1	.0053
		1980 Co[1]	0	3	.0112	0.1	2.8	.0138	40.5	.0002
								Total	1916.0	.0243

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[1] Time interval between mean date of egg deposition for cutthroat (4/3/80) and coho (11/28/79) and sampling date (7/26/79) is 105 and 233 days, respectively.

Appendix Table 4. Estimated population size, average weight, production, and associated parameters of salmonids at electrofishing sites on Kelsey Creek (continued).

Dates (length)	S.A. (m^2)	Year class-species	Population			Weight			Production	
			N_1	N_2	Z(1/days)	$\overline{W}_1$(gms)	$\overline{W}_2$(gms)	g(1/days)	P(gms)	P/m^2day
7/18/80-09/20/80	870.1	1980 Ct	144	144	.0000	1.2	4.1	.0188	461.1	.0083
(64)		1979 Ct	49	30	.0074	23.5	29.7	.0037	242.0	.0043
		1978 Ct	3	5	-.0071	69.0	68.4	-.0001	0	0
		1980 Co	3	3	0	2.9	9.2	.0179	20.8	.0004
								Total	723.9	.0130
9/20/80-11/11/80	870.1	1980 Ct	144	83	.0106	4.1	5.6	.0062	171.5	.0038
(52)		1979 Ct	30	27	.0026	29.7	33.4	.0022	103.8	.0023
		1978 Ct	5	3	.0098	68.4	92.3	.0058	92.9	.0020
		1980 Co	3	0		9.2	14.2	.0084	6.0	.0001
								Total	374.3	.0083
11/11/80-12/25/80	870.1	1979 Ct	83	99	-.0037	5.6	7.8	.0074	200.9	.0052
(44)		1979 Ct	27	19	.0076	33.4	44.4	.0065	246.2	.0064
		1978 Ct	3	1	.0250	92.3	102.7	.0024	20.1	.0005
		1980 Co	0	3		14.2	18.5	.0060	7.3	.0002
								Total	474.5	.0124
12/25/80-04/06/81	870.1	1980 Ct	99	140	-.0035	7.8	17.7	.0080	1338.3	.0151
(102)		1979 Ct	19	22	-.0015	44.4	64.5	.0037	422.8	.0048
		1978 Ct	1	0		102.7	126.6	.0039	10.8	.0001
		1981 Co	0	40		0.2	1.0	.0179	36.5	.0004
		1980 Co	3	4	-.0028	18.5	16.2	-.0013	0	0
								Total	1808.3	.0204
4/06/81-05/25/81	924.2	1981 Ct	0	698		0.1	0.4	.0203	131.6	.0029
(49)		1980 Ct	140	131	.0014	17.7	27.7	.0091	1365.4	.0301
		1979 Ct	22	9	.0183	64.5	80.8	.0046	242.6	.0054
		1981 Co	40	46	-.0025	1.0	4.2	.0296	168.6	.0037
		1980 Co[1]	4	0		16.2	19.8	.0041	6.5	.0001
								Total	1914.8	.0423
5/25/81-07/04/81	924.2	1981 Ct	698	560	.0055	0.4	1.3	.0300	581.7	.0157
(40)		1980 Ct	131	82	.0118	27.7	27.2	-.0004	0	0
		1979 Ct	9	3	.0260	80.8	91.8	.0032	65.0	.0018
		1981 Co	46	27	.0131	4.2	6.4	.0105	98.0	.0026
								Total	744.7	.0201
7/04/81-08/16/81	924.2	1981 Ct	560	479	.0036	1.3	4.1	.0277	1604.5	.0404
(43)		1980 Ct	82	59	.0076	27.2	29.1	.0015	130.2	.0033
		1979 Ct	3	3	.0014	91.8	103.3	.0028	35.6	.0009
		1981 Co	27	14	.0152	6.4	7.6	-.0040	32.7	.0008
								Total	1803.0	.0454

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[1] Outmigration estimated to be complete on June 18, based on data from downstream trap in 1981.

Appendix Table 5. Estimated population size, average weight, production, and associated parameters of salmonids at electrofishing sites on Bear Creek.

			Population			Weight			Production	
Dates (length)	S.A. (m^2)	Year class-species	N_1	N_2	Z(1/days)	W_1(gms)	W_2(gms)	g(1/days)	p(gms)	p/m^2day
6/25/79-07/23/79 (28)	793.1	1978 Ct	47	22	.0260	11.9	12.3	.0011	17.8	.0008
		1977 Ct	27	5	.0609	27.1	27.2	.0002	2.2	.0001
		1976 Ct	3	3	-.0103	123.3	154.3	.0080	75.5	.0034
		1979 Co	254	204	.0076	2.3	2.3	.0009	15.5	.0007
								Total	111.0	.0050
7/23/79-09/22/79 (61)	793.1	1979 Ct	20	26	-.0046	1.7	2.5	.0064	14.5	.0003
		1978 Ct	22	12	.0102	12.3	12.0	-.0004	0	0
		1977 Ct	5	6	-.0022	27.2	34.0	.0037	29.0	.0006
		1976 Ct	3	1	.0227	154.3	174.0	.0020	58.1	.0012
		1979 Co	204	160	.0040	2.3	2.9	.0035	91.9	.0019
								Total	193.5	.0040
9/22/79-12/12/79 (81)	793.1	1979 Ct	26	2	.0346	2.5	5.5	.0099	45.0	.0007
		1978 Ct	12	7	.0063	12.0	17.3	.0045	51.4	.0008
		1977 Ct	6	5	.0036	34.0	52.0	.0052	89.9	.0014
		1979 Co	160	58	.0122	2.9	5.0	.0068	244.1	.0038
								Total	430.4	.0067
12/12/79-05/06/80 (146)	793.1	1979 Ct	2	22	-.0178	5.5	11.7	.0052	69.5	.0006
		1978 Ct	7	5	.0017	17.3	27.2	.0031	57.9	.0005
		1977 Ct	5	4	.0028	52.0	66.6	.0017	69.5	.0006
		1980 Co[1]	0	370	.0052	0.1	0.8	.0119	186.5	.6014
		1979 Co	58	27	.0018	5.0	8.8	.0039	162.1	.0014
								Total	545.5	.0045
5/06/80-07/21/80 (76)	793.1	1980 Ct[2]	0	112	.0041	0.1	1.8	.0300	42.8	.0005
		1979 Ct	22	17	.0021	11.7	12.8	.0012	24.1	.0004
		1978 Ct	5	10	-.0059	27.2	28.4	.0006	6.0	.0001
		1977 Ct	4	4	-.0053	66.6	73.7	.0013	24.1	.0004
		1980 Co	370	283	.0036	0.8	2.4	.0137	307.4	.0051
								Total	404.4	.0065

[1] Time interval between mean date of coho egg deposition (11/19/79) and sampling date (5/6/80) is 168 days.

[2] Time interval between mean date of cutthroat egg deposition (4/3/80) and sampling date (7/21/80) is 108 days.

Appendix Table 5. Estimated population size, average weight, production, and associated parameters at electrofishing sites on Bear Creek (continued).

Dates (length)	S.A. (m^2)	Year class-species	Population			Weight			Production	
			N_1	N_2	Z(1/days)	W_1(gms)	W_2(gms)	g(1/days)	p(gms)	p/m^2day
7/21/80-09/24/80	793.1	1980 Ct	112	123	-.0015	1.8	2.7	.0068	116.5	.0023
(65)		1979 Ct	17	28	-.0074	11.2	14.7	.0042	80.8	.0016
		1978 Ct	10	11	-.0004	27.1	25.6	-.0009	0	0
		1977 Ct	4	5	-.0034	78.6	83.4	.0009	22.5	.0004
		1976 Ct	1	1	0	110.0	108.0	-.0003	0	0
		1980 Co	283	229	.0033	1.8	3.0	.0041	179.4	.0035
								Total	399.166	.0077
9/24/80-11/08/80	793.1	1980 Ct	123	93	.0063	2.7	4.6	.0118	203.5	.0064
(45)		1979 Ct	28	15	.0136	14.7	14.6	-.0001	0	0
		1978 Ct[2]	11	4	.0225	25.6	37.5	.0085	80.9	.0023
		1977 Ct	5	3	.0114	83.4	87.7	.0011	17.0	.0005
		1976 Ct	1	1	0	108.0	233.0	.0171	[3]	
		1980 Co	226	167	.0068	3.0	4.2	.0072	222.4	.0062
								Total	523.73	.0147
11/08/80-01/05/81	793.1	1980 Ct	93	103	.0018	4.6	4.7	.0002	6.0	.0001
(58)		1979 Ct	15	27	-.0099	14.6	13.2	-.0017	0	0
		1978 Ct	4	5	-.0038	37.5	36.4	-.0005	0	0
		1977 Ct	3	2	.0158	87.7	72.5	-.0033	0	0
		1976 Ct	1	0		233.0			[3]	
		1981 Co	0	8		0.2	0.4	.0169	1.7	<.0001
		1980	167	76	.0136	4.2	3.9	-.0010	0	0
								Total	7.62	.0002
1/05/81-03/29/81	793.1	1980 Ct	103	92	.0014	4.7	9.4	.0084	468.4	.0071
(83)		1979 Ct	27	7	.0155	13.2	25.8	.0080	180.7	.0027
		1978 Ct	5	6	-.0029	36.4	56.1	.0052	116.7	.0017
		1977 Ct	2	2	-.0010	72.5	113.5	.0054	87.9	.0013
		1981 Co	8	514	-.0501	0.4	0.5	.0028	32.3	.0005
		1980 Co	76	90	-.0020	3.9	6.4	.0058	211.7	.0032
								Total	1097.71	.0167
3/29/81-05/16/81	793.1	1981 Ct	0	180		0.1	0.4	.0023	42.5	.0011
(48)		1980 Ct	92	91	<.0001	9.4	11.5	.0041	188.2	.0049
		1979 Ct	7	17	-.0185	25.8	23.6	-.0019	0	0
		1978 Ct	6	1	.0373	56.1	50.5	-.0022	0	0
		1977 Ct	2	2	.0018	113.5	134.5	.0035	43.8	.0012
		1981 Co	514	351	.0079	0.5	1.5	.0214	412.4	.0108
		1980 Co	90	69	.0055	6.4	7.7	.0039	105.0	.0028
								Total	791.9	.0208
5/16/81-06/29/81	793.1	1981 Ct	180	191	-.0014	0.4	1.0	.0193	113.6	.0033
(44)		1980 Ct	91	71	.0058	11.5	12.6	.0021	87.9	.0025
		1979 Ct	17	8	.0173	23.6	31.3	.0064	92.7	.0027
		1978 Ct	1	2	-.0158	50.5	58.2	.0032	11.9	.0003
		1977 Ct	2	2	0	134.5	126.0	-.0015	0	0
		1981 Co	351	403	-.0031	1.5	2.4	.0106	348.5	.0100
		1980 Co[1]	69	0		7.7	8.9	.0028	24.4	.0007
								Total	679.0	.0195
6/29/81-08/23/81	793.1	1981 Ct	191	122	.0081	1.0	2.3	.0153	199.9	.0046
(55)		1980 Ct	71	56	.0042	12.6	11.8	-.0011	0	0
		1979 Ct	8	8	.0001	31.3	31.4	.0001	1.2	<.0001
		1978 Ct	2	3	-.0074	58.2	61.9	.0011	9.1	.0002
		1977 Ct	2	1	.0013	126.0	90.3	-.0061	0	0
		1981 Co	403	315	.0045	2.4	2.6	.0019	96.1	.0022
								Total	306.3	.0070

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[1] Outmigration estimated to be complete on June 18, based on data from operation of downstream trap on Kelsey Creek in 1981.

[2] Three apparent hatchery rainbow trout plants not included.

[3] This single large fish was not used in production calculations.

Appendix Table 6. The estimated number of immigrants and survival of the 1979 year class of cutthroat trout at sampling sites in Kelsey Creek for the period April 1980 to August 1981, and the variance of these estimates.

Site	Species Age class	Time period	Survival		Immigrants	
			$\hat{\theta}$	$V(\hat{\theta})$	$\hat{I}$	$V(\hat{I})$
KC1	1979 Ct.	7/17/80- 9/16/80	.33	.015	[1]	
		9/16/80-11/11/80	.31	.067	7.0	.48
KC4	1979 Ct.	11/11/80-12/20/80	.50	.066	1.3	.60
		12/20/80- 4/07/81	.80	.149	4.6	.60
KC5	1979 Ct.	7/19/80- 9/24/80	.62	.020	5.9	1.00
		9/24/80-11/11/80	.53	.027	3.5	.59
		11/11/80-12/23/80	.31	.047	1.3	.58

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[1] Number of immigrants estimated to be negative.

Appendix Table 7. The estimated number of immigrants and survival of the 1980 year class of cutthroat trout at sampling sites in Kelsey Creek for the period April 1980 to August 1981, and the variance of these estimates.

Site	Species Age class	Time period	Survival		Immigrants	
			$\hat{\theta}$	$V(\hat{\theta})$	$\hat{I}$	$V(\hat{I})$
KC1	1980 Ct.	7/17/80- 9/16/80	.75	.130	19.6	2.42
		9/16/80-11/11/80	.60	.023	7.6	.51
		11/11/80- 1/10/81	.82	.039	2.5	2.75
		1/10/81- 4/02/81	.56	.024	6.9	.44
		4/02/81- 5/12/81	.70	.027	9.4	.46
		5/12/81- 6/26/81	.60	.022	5.3	.47
KC2	1980 Ct.	7/17/80- 9/17/80	.62	.060	43.4	828.30
		9/17/80-11/10/80	.44	.005	2.8	.46
		11/10/80-12/23/80	.87	.011	14.7	1.13
		12/23/80- 4/07/81	.50	.008	29.8	30.68
		4/07/81- 5/30/81	.46	.007	6.7	6.72
		5/30/81- 7/10/81	.71	.008	4.9	.22
KC3	1980 Ct.	7/19/80- 9/23/80	.67	.076	24.8	1.67
		9/23/80-11/11/80	.54	.012	8.1	.36
		11/11/80-12/19/80	1.03	.024	[1]	
		12/19/80- 4/06/81	.71	.016	14.6	1.19
		4/06/81- 5/23/81	.62	.013	2.2	.39
		5/23/81- 7/02/81	.50	.012	2.0	.24
KC4	1980 Ct.	11/11/80-12/20/80	.40	.175	8.5	.90
		12/20/80- 4/07/81	.36	.051	9.4	1.12
		4/07/81- 5/28/81	.62	.327	[1]	
		5/28/81- 7/02/81	.25	.068	7.8	.59
KC5	1980 Ct.	9/24/80-11/11/80	.72	.031	[1]	
		11/11/80-12/23/80	1.25	.366	[1]	
		12/23/80- 4/09/81	.48	.059	29.3	4.07
		4/09/81- 5/30/81	.53	.025	36.3	1.82
		5/30/81- 7/10/81	.25	.004	4.4	.22

[1] Number of immigrants estimated to be negative.

Appendix Table 8. The estimated number of immigrants and survival of the 1981 year class of cutthroat trout at sampling sites in Kelsey Creek for the period April 1980 to August 1981, and the variance of these estimates.

Site	Species Age class	Time period	Survival		Immigrants	
			$\hat{\theta}$	$V(\hat{\theta})$	$\hat{I}$	$V(\hat{I})$
KC2	1981 Ct.	5/30-7/10/81	.56	.060	126.7	13.44
KC5	1981 Ct.	5/30-7/10/81	.36	.008	24.9	.75

Appendix Table 9. The estimated number of immigrants and survival of the 1981 year class of coho salmon at sampling sites in Kelsey Creek for the period April 1980 to August 1981, and the variance of these estimates.

Site	Species Age class	Time period	Survival		Immigrants	
			$\hat{\theta}$	$V(\hat{\theta})$	$\hat{I}$	$V(\hat{I})$
KC3	1981 Co.	4/06-5/23/81	.39	.017	12.4	.60
		5/23-7/02/81	.53	.038	1.2	1.00

Appendix Table 10. The estimated number of immigrants and survival of the 1979 year class of cutthroat trout at sampling sites in Bear Creek for the period May 1980 to August 1981, and the variance of these estimates.

			Survival		Immigrants	
Site	Species Age class	Time period	$\hat{\theta}$	$V(\hat{\theta})$	$\hat{I}$	$V(\hat{I})$
BC3	1979 Ct.	7/22- 9/23/80	.90	.105	12.5	46.97
		9/23-11/07/80	1.15	1.262	[1]	

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[1] Number of immigrants estimated to be negative.

Appendix Table 11. The estimated number of immigrants and survival of the 1980 year class of cutthroat trout at sampling sites in Bear Creek for the period May 1980 to August 1981, and the variance of these estimates.

			Survival		Immigrants	
Site	Species Age class	Time period	$\hat{\theta}$	$V(\hat{\theta})$	$\hat{I}$	$V(\hat{I})$
BC1	1980 Ct.	11/09/80-12/21/80	.71	.078	14.4	.47
		12/21/80- 4/04/80	.37	.018	7.3	.36
		4/04/81- 5/23/81	.62	.034	11.6	5.54
		5/23/81- 7/01/81	.61	.020	11.0	2.31
BC2	1980 Ct.	11/08/80- 1/12/81	1.00	.085	9.2	1.31
		1/12/81- 3/26/81	.58	.027	9.5	1.32
		3/26/81- 5/16/81	.63	.025	11.8	.72
		5/16/81- 6/29/81	.84	.043	[1]	
BC3	1980 Ct.	9/23/80-11/07/80	.66	.011	41.6	188.20
		11/07/80- 1/12/81	.86	.023	2.9	123.45
		1/12/81- 3/25/81	.72	.022	10.4	31.41
		3/25/81- 5/10/81	.51	.009	17.6	22.88
		5/10/81- 6/28/81	.64	.008	.8	6.32

[1] Number of immigrants estimated to be negative.

Appendix Table 12. The estimated number of immigrants and survival of the 1981 year class of cutthroat trout at sampling sites in Bear Creek for the period May 1980 to August 1981, and the variance of these estimates.

Site	Species Age class	Time period	Survival		Immigrants	
			$\hat{\theta}$	$V(\hat{\theta})$	$\hat{I}$	$V(\hat{I})$
BC2	1981 Ct.	5/16-6/29/81	.15	.015	49.9	62.34
BC3	1981 Ct.	5/10-6/28/81	.19	.009	108.8	11.22

Appendix Table 13. The estimated number of immigrants and survival of the 1980 year class of coho salmon at sampling sites in Bear Creek for the period May 1980 to August 1981, and the variance of these estimates.

Site	Species Age class	Time period	Survival $\hat{\theta}$	Survival $V(\hat{\theta})$	Immigrants $\hat{I}$	Immigrants $V(\hat{I})$
BC1	1980 Co.	7/20/80- 9/27/80	.27	.012	54.0	14.63
		9/27/80-11/09/80	.58	.096	5.1	44.78
		11/09/80-12/21/80	.66	.111	17.6	19.24
BC2	1980 Co.	7/21/80- 9/21/80	.91	.129	37.0	147.84
		9/21/80-11/08/80	.24	.031	34.9	14.96
		11/08/80- 1/12/81	.30	.035	2.3	6.01
BC3	1980 Co.	7/22/80- 9/23/80	.37	.167	46.2	13.13
		9/23/80- 11/07/80	.33	.098	55.7	1129.09
		11/07/80- 1/12/81	.09	.005	7.1	525.29

Appendix Table 14. The estimated number of immigrants and survival of the 1981 year class of coho salmon at sampling sites in Bear Creek for the period May 1980 to August 1981, and the variance of these estimates.

Site	Species Age class	Time period	Survival		Immigrants	
			$\hat{\theta}$	$V(\hat{\theta})$	$\hat{I}$	$V(\hat{I})$
BC1	1981 Co.	4/04-5/23/81	.21	.002	65.0	2.34
		5/23-7/01/81	.54	.008	72.8	42.82
BC2	1981 Co.	3/26-5/16/81	.21	.001	104.6	26.67
		5/16-6/29/81	.59	.008	98.6	10.12
BC3	1981 Co.	3/25-5/10/81	.49	.007	51.9	.62
		5/10-6/28/81	.43	.003	48.2	15.26

Appendix Table 15. The estimated abundance of the 1978 year class of cutthroat trout at sampling sites on Kelsey Creek in the period 4/80 to 8/81, and the variance of this estimate. (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area.

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
KC1	1978 Ct.	4/24/80	0	—					153.9	0
		7/17/80	2.2	.02						.014
		9/17/80	2.0	.00						.013
		11/11/80	1.0[1]	.00						.006
		1/10/81	1.0[1]	.00						.006
KC2	1978 Ct.	4/26/80	5.8	19.20					120.5	.005
KC3	1978 Ct.	4/26/80	2.0	0					212.5	.009
KC4	1978 Ct.	4/27/80	2.2	.02					170.8	.229
		7/17/80	0	—						0
		9/18/80	1.0[1]	0						.006
		11/11/80	2.0	0						.012
KC5	1978 Ct.	4/27/80	5.9	.90					212.5	.028
		7/19/80	1.0[1]	0						.005
		9/24/80	1.0[1]	0						.005

[1] One fish captured on first pass. $\hat{N}_E$ set equal to 1, $V(\hat{N}_E)$ set equal to 0.

Appendix Table 16. The estimated abundance of the 1979 year class of cutthroat trout at sampling sites on Kelsey Creek in the period 4/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area.

Site	Species Age Class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
KC1	1979 Ct.	4/24/80	31.4	.01					153.9	.204
		7/17/80	22.2	<.01						.144
		9/16/80	7.1	<.01	10.1	5.88	7.1	<.01		.046
		11/11/80	9.2	.01	10.0	16.46	9.2	.01		.060
		1/10/81	7.0	0						.045
		4/02/81	1.0^1	0						.006
		5/30/81	2.0	0						.013
		6/28/81	2.2	.02						.014
		8/14/81	1.0^1	0						.006
KC2	1979 Ct.	4/26/80	43.8	2.56					120.5	.364
		7/17/80	7.0	<.01						.058
		9/17/80	5.0	<.01						.042
		11/10/81	0	—						0
		12/23/80	2.0	0						.017
		4/17/81	6.0	<.01						.050
		5/30/81	3.0	0						.025
KC3	1979 Ct.	4/26/80	28.6	.86					212.5	.135
		7/19/80	3.0	0						.014
		9/17/80	2.0	0						.009
		11/11/80	0	—						0
		12/19/80	1.0^1	0						.005
		4/06/81	1.0^1	0					205.8	.005
		5/23/81	0	—						0
		7/02/81	0	—						0
		8/14/81	1.0^1	0						.005

Appendix Table 16. The estimated abundance of the 1979 year class of cutthroat trout at sampling sites on Kelsey Creek in the period 4/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area. (Continued).

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
KC4	1979 Ct.	4/27/80	39.2	<.01					170.8	.229
		7/17/80	0	—						0
		9/18/80	0	—						0
		11/11/80	5.3	.99						.031
		12/20/80	4.0	0	5.0	1.69	4.0	0		.023
		4/07/81	7.0	0	4.0	2.67	7.0	0		.030
		5/28/81	1.0[2]	1.0						.004
		7/02/81	1.0[1]	0						.004
KC5	1979 Ct.	4/27/80	30.5	.03					212.5	.144
		7/19/80	16.7	1.71						.079
		9/24/80	16.3	.01	17.0	2.58	16.3	.01		.077
		11/11/80	12.1	.15	13.0	5.19	12.1	.15		.057
		12/23/80	5.0	0	6.0	7.87	5.0	0		.023
		4/09/81	7.0	<.01						.033
		5/30/81	3.0	0						.014
		7/10/81	0	—						0
		8/18/81	1.0[1]	0						.005

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[1] One fish captured on first pass. $\hat{N}_E$ set equal to 1, $V(\hat{N}_E)$ set equal to 0.

[2] Catch-effort population estimate not possible. $\hat{N}_E$, $V(\hat{N}_E)$ set equal to total catch.

Appendix Table 17. The estimated abundance of the 1980 year class of cutthroat trout at sampling sites on Kelsey Creek in the period 4/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area.

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
KC1	1980 Ct.	4/24/80	0	—					153.9	0
		7/17/80	18.4	.03						.120
		9/16/80	22.2	<.01	23.0	17.26	22.2	<.01		.144
		11/11/80	19.1	<.01	26.2	20.53	19.1	<.01		.124
		1/10/81	17.8	2.05	36.0	97.15	18.2	2.01		.118
		4/02/81	17.0	0	21.3	12.59	17.0	0		.110
		5/30/81	21.2	<.01	30.1	38.68	21.2	<.01		.138
		6/28/81	18.0	<.01	22.5	14.84	18.0	<.01		.117
		8/14/81	8.0	0						.052
KC2	1980 Ct.	4/26/80	8.0	.93					120.5	.007
		7/17/80	106.7	2137.82						.885
		9/17/80	80.2	.12	80.0	44.74	80.2	.12		.665
		11/10/80	32.0	.04	48.6	30.04	32.1	.04		.266
		12/23/80	41.5	.76	53.8	48.25	41.7	.75		.346
		4/17/81	59.7	392.42	47.4	32.67	48.3	30.16		.401
		5/30/81	28.0	<.01	34.7	7.14	28.0	<.01		.233
		7/10/81	24.0	<.01	26.5	1.38	24.0	<.01		.200
		8/18/81	23.0	0						.191
KC3	1980 Ct.	4/26/80	1.0[2]	1.00					212.5	.005
		7/19/80	18.2	.64						.086
		9/17/80	29.0	<.01	30.0	16.43	29.0	<.01		.136
		11/11/80	23.3	.01	34.3	17.71	23.3	.01		.109
		12/19/80	18.7	1.76	30.9	21.58	19.7	1.63		.093
		4/06/81	28.6	.05	33.1	19.22	28.6	.05	205.8	.139
		5/23/81	20.0	0	22.4	1.03	20.0	0		.097
		7/02/81	12.0	<.01	13.0	.46	12.0	<.01		.058
		8/14/81	12.0	0						.058

Appendix Table 17. The estimated abundance of the 1980 year class of cutthroat trout at sampling sites on Kelsey Creek in the period 4/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area. (Continued).

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
KC4	1980 Ct.	4/27/80	1.0[2]	1.0					170.8	.006
		7/17/80	1.0[1]	0						.006
		9/18/80	3.1	<.01						.018
		11/11/80	4.0	0						.023
		12/20/80	10.1	.20	11.0	23.12	10.1	.20		.059
		4/17/81	13.1	.61	13.0	10.04	13.1	.58	231.6	.056
		5/28/81	7.0	0	16.0	209.14	7.0	0		.030
		7/12/81	9.5	.11	11.0	20.13	9.6	.11		.041
		8/15/81	2.0	0						.009
KC5	1980 Ct.	4/27/80	0	—					212.5	0
		7/19/80	0	—						0
		9/24/80	10.0	<.01						.047
		11/11/80	5.0	0	7.2	.79	5.00	0		.024
		12/23/80	9.1	.24	35.0	655.54	9.15	.24		.043
		4/09/81	34.1	4.16	32.0	20.76	33.73	3.47		.159
		5/30/81	54.1	<.01	71.6	336.89	54.13	<.01		.255
		7/10/81	18.1	<.01	19.0	.80	18.14	<.01		.085
		8/18/81	14.0	<.01						.066

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[1] One fish captured on first pass. $\hat{N}_E$ set equal to 1, $V(\hat{N}_E)$ set equal to 0.

[2] Catch-effort population estimate not possible. $\hat{N}_E$, $V(\hat{N}_E)$ set equal to total catch.

Appendix Table 18. The estimated abundance of the 1981 year class of cutthroat trout at sampling sites on Kelsey Creek in the period 4/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area.

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
KC1	1981 Ct.	5/30/81	3.0[1]	3.00					153.9	.019
		6/28/81	17.2	<.01						.111
		8/14/81	23.3	.01						.151
KC2	1981 Ct.	5/30/81	92.7	2.44					120.5	.769
		7/10/81	133.8	7.74	128.0	88.74	133.2	7.12		1.106
		8/18/81	103.4	<.01						.858
KC3	1981 Ct.	5/23/81	224.8	2.24					205.8	1.092
		7/02/81	179.8	6.34						.874
		8/14/81	138.9	.03						.675
KC4	1981 Ct.	5/28/81	331.1	6.01					231.6	1.430
		7/02/81	189.4	27.04						.818
		8/15/81	131.4	<.01						.567
KC5	1981 Ct.	5/30/81	46.0	2.86					212.5	.217
		7/10/81	39.5	.01	46.1	42.33	39.45	.01		.186
		8/18/81	82.0	.07						.386

[1] Catch effort population estimate not possible. $\hat{N}_E$, $V(\hat{N}_E)$ set equal to total catch.

Appendix Table 19. The estimated abundance of the 1979 year class of coho salmon at sampling sites on Kelsey Creek in the period 4/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area.

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
KC1	1979 Co.	4/24/80	0	—					153.9	0
KC2	1979 Co.	4/26/80	11.7	2.99					120.5	.097
KC3	1979 Co.	4/26/80	13.3	.01					212.5	.062
KC4	1979 Co.	4/27/80	11.2	<.01					170.8	.065
KC5	1979 Co.	4/27/80	3.0	0					212.5	.014

Appendix Table 20. The estimated abundance of the 1980 year class of coho salmon at sampling sites on Kelsey Creek in the period 4/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area.

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
KC1	1980 Co.	4/24/80	5.0	0					153.9	.032
		7/17/80	2.0	0						.013
		9/16/80	2.0	0						.013
KC2	1980 Co.	4/24/80	1.0^1	0					120.5	.008
		7/17/80	0	—						0
		9/17/80	0	—						0
		11/10/80	0	—						0
		12/23/80	0	—						0
		4/17/81	2.0	0						.017
KC3	1980 Co.	4/26/80	2.0	0					212.5	.009
		7/19/80	1.0^2	1.0						.005
		9/17/80	1.0	0						.005
KC4	1980 Co.	4/27/80	4.0^2	4.00					170.8	.023
		7/17/80	0	—						0
		9/18/80	0	—						0
		11/11/80	0	—						0
		12/20/80	1.0^1	0						.006
		4/07/81	2.0^1	0					231.6	.009

Appendix Table 20. The estimated abundance of the 1980 year class of coho salmon at sampling sites on Kelsey Creek in the period 4/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area. (Continued).

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
KC5	1980 Co.	4/27/80	0	—					212.5	0
		7/19/80	0	—						0
		9/24/80	0	—						0
		11/11/80	0	—						0
		12/23/80	2.0[2]	2.00						.009

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[1] One fish captured on first pass. $\hat{N}_E$ set equal to 1, $V(\hat{N}_E)$ set equal to 0.

[2] Catch-effort population estimate not possible. $\hat{N}_E$, $V(\hat{N}_E)$ set equal to total catch.

Appendix Table 21. The estimated abundance of the 1981 year class of coho salmon at sampling sites on Kelsey Creek in the period 4/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area.

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
KC1	1981 Co.	4/02/81	0	—					153.9	0
		5/30/81	1.0[1]	0						.006
		6/28/81	4.0	0						.026
KC2	1981 Co.	4/17/81	1.0[2]	1.00					120.5	.008
		5/30/81	4.4	.08						.036
		7/10/81	1.0[1]	0						.008
KC3	1981 Co.	4/06/81	35.3	<.01					205.8	.172
		5/25/81	26.2	<.01	43.8	171.76	26.2	<.01		.127
		7/02/81	15.0	0	25.3	69.73	15.0	0		.073
		8/14/81	8.0	<.01						.039
KC4	1981 Co.	4/07/81	3.0[2]	3.00					231.6	.013
		5/28/81	13.1	.61						.056
		7/02/81	6.0	6.00						.026
		8/15/81	5.0	<.01						.022
KC5	1981 Co.	4/09/81	1.0[2]	1.00					212.5	.005
		5/30/81	1.0[2]	1.00						.005
		7/10/81	1.0[1]	0						.005
		8/18/81	1.0[1]	0						.005

[1] One fish captured on first pass. $\hat{N}_E$ set equal to 1, $V(\hat{N}_E)$ set equal to 0.

[2] Catch-effort population estimate not possible. $\hat{N}_E$, $V(\hat{N}_E)$ set equal to total catch.

Appendix Table 22. The estimated abundance of the 1976 year class of cutthroat trout at sampling sites on Bear Creek in the period 5/80 to 8/81, and the variance of the estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area.

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
BC1	1976 Ct.	All dates	None captured							
BC2	1976 Ct.	All dates	None captured							
BC3	1976 Ct.	5/06/80	0	—					251.3	0
		7/22/80	1.0[1]	0						.004
		9/22/80	1.0[1]	0						.004
		11/07/80	1.0[1]	0						.004

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[1] One fish captured on first pass. $\hat{N}_E$ set equal to 1, $V(\hat{N}_E)$ set equal to 0.

Appendix Table 23. The estimated abundance of the 1977 year class of cutthroat trout at sampling sites on Bear Creek in the period 5/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area.

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
BC1	1977 Ct.	5/04/80	1.0[1]	0					247.2	.004
		7/20/80	0	—						0
		9/27/80	1.0[2]	1.00						.004
BC2	1977 Ct.	5/06/80	2.0[2]	2.00					294.5	.007
		7/21/80	2.2	.02						.007
		9/21/80	2.0	0						.007
		11/08/80	2.0[2]	2.00						.007
		1/12/80	1.0[1]	0						.003
		3/26/80	2.2	.02						.007
		5/16/81	2.0	0						.007
		6/29/81	2.0	0						.007
		8/17/81	1.0[1]	0						.003
BC3	1977 Ct.	5/06/80	0	0					251.3	0
		7/22/80	2.2	.02						.009
		9/22/80	2.0	0						.008
		11/07/80	1.0[1]	0						.004
		1/12/80	1.0[1]	0						.004

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[1] One fish captured on first pass. $\hat{N}_E$ set equal to 1, $V(\hat{N}_E)$ set equal to 0.

[2] Catch-effort population estimate not possible. $\hat{N}_E$, $V(\hat{N}_E)$ set equal to total catch.

Appendix Table 24. The estimated abundance of the 1978 year class of cutthroat trout at sampling sites on Bear Creek in the period 5/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area.

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
BC1	1978 Ct.	5/04/80	2.2	.02					247.2	.009
		7/20/80	2.2	.02						.009
		9/27/80	1.0[2]	1.00						.004
		11/09/80	0	—						0
		12/21/80	1.0[1]	0						.004
BC2	1978 Ct.	5/06/80	1.0[1]	0					294.5	.003
		7/21/80	2.2	.02						.007
		9/21/80	3.8	1.50						.013
		11/08/80	3.0[2]	3.00						.010
		1/12/80	0	—						0
		3/26/81	2.0	0						.007
		5/16/81	1.0[1]	0						.003
		6/29/81	1.0[1]	0						.003
		8/27/81	1.0[1]	0						.003

Appendix Table 24. The estimated abundance of the 1978 year class of cutthroat trout at sampling sites on Bear Creek in the period 5/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area. (Continued).

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
BC3	1978 Ct.	5/06/80	2.0[2]	2.00[2]					251.3	.008
		7/22/80	6.0	<.01						.024
		9/22/80	5.9	.90						.023
		11/07/80	4.0	0						.016
		1/12/81	4.0	0						.016
		3/25/81	4.4	.08						.017
		5/10/81	0	—						0
		6/28/81	1.0	0						.004
		8/26/81	2.0	0						.008

[1] One fish captured on first pass. $\hat{N}_E$ set equal to 1, $V(\hat{N}_E)$ set equal to 0.

[2] Catch-effort population estimate not possible. $\hat{N}_E$, $V(\hat{N}_E)$ set equal to total catch.

Appendix Table 25. The estimated abundance of the 1979 year class of cutthroat trout at sampling sites on Bear Creek in the period 5/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area.

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
BC1	1979 Ct.	5/04/80	0	—					247.2	0
		7/20/80	4.0	<.01						.016
		9/27/80	5.9	.90						.024
		11/09/80	5.9	.90						.024
		12/21/80	3.0	0						.012
		4/04/81	2.0	0						.008
		5/23/81	5.8	19.17						.024
		7/01/81	1.0[2]	1.00						.004
		8/27/81	2.0	0						.008
BC2	1979 Ct.	5/06/80	7.6	4.80					294.5	.026
		7/21/80	5.8	19.17						.020
		9/21/80	3.0	0						.010
		11/08/80	1.0[1]	0						.003
		1/12/80	5.0	<.01						.017
		3/26/81	4.4	.08						.015
		5/16/81	7.0	0						.024
		6/29/81	5.0	<.01						.017
		8/17/81	3.0	0						.010

Appendix Table 25. The estimated abundance of the 1979 year class of cutthroat trout at sampling sites on Bear Creek in the period 5/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area. (Continued).

Site	Species Age Class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
BC3	1979 Ct.	5/04/80	13.5	.06					251.3	.054
		7/22/80	7.1	<.01						.028
		9/22/80	22.6	691.63	18.6	49.53	18.9	46.22		.075
		11/07/80	8.1	<.01	39.4	1618.59	8.1	<.01		.032
		1/12/81	18.5	15461.50						.074
		3/25/81	1.0[1]	0						.004
		5/10/81	4.4	.08						.017
		6/28/81	2.0	0						.008
		8/26/81	3.0	0						.012

[1] One fish captured on first pass. $\hat{N}_E$ set equal to 1, $V(\hat{N}_E)$ set equal to 0.

[2] Catch-effort population estimate not possible. $\hat{N}_E$, $V(\hat{N}_E)$ set equal to total catch.

Appendix Table 26. The estimated abundance of the 1980 year class of cutthroat trout at sampling sites on Bear Creek in the period 5/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area.

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
BC1	1980 Ct.	7/20/80	25.7	151.04					247.2	.104
		9/27/80	32.7	5185.78						.132
		11/09/80	6.0	.00						.024
		12/21/80	18.7	.12	19.0	11.52	18.7	.12		.076
		4/04/81	14.2	<.01	15.0	3.63	14.2	.01		.057
		5/23/81	19.7	5.66	26.4	46.54	20.4	5.05		.083
		7/01/81	23.4	.02	24.0	5.32	23.4	.02		.095
		8/27/81	14.0	<.01						.057
BC2	1980 Ct.	7/20/80	23.0	.28					294.5	.078
		9/27/80	10.9	.44						.037
		11/09/80	6.1	0						.021
		12/21/80	15.2	.80	26.2	102.81	15.3	.79		.052
		4/04/81	18.2	.64	31.9	95.18	18.3	.64		.062
		5/23/81	23.3	.01	24.0	2.89	23.3	.01		.079
		7/01/81	16.0	<.01	26.5	40.30	16.0	<.01		.054
		8/27/81	15.1	<.01						.051

Appendix Table 26. The estimated abundance of the 1980 year class of cutthroat trout at sampling sites on Bear Creek in the period 5/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area. (Continued).

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
BC3	1980 Ct.	7/22/80	63.0	2.00					251.3	.251
		9/22/80	79.4	53.74						.316
		11/07/80	72.4	768.66	82.6	208.36	80.4	163.92		.320
		1/12/81	75.4	.37	92.4	322.03	45.5[1]	.37		.274
		3/25/81	56.5	31.73	109.5	637.16	59.0	30.22		.235
		5/10/81	45.3	19.39	54.8	57.30	47.7	14.49		.190
		6/28/81	31.2	<.01	38.1	9.34	31.2	<.01		.124
		8/26/81	27.0	<.01						.107

[1] $\hat{N}_E$ and $\hat{N}_{MR}$ differ significantly. Unweighted mean of estimates employed to estimate $\hat{N}_P$.

Appendix Table 27. The estimated abundance of the 1981 year class of cutthroat trout at sampling sites on Bear Creek in the period 5/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area.

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
BC1	1981 Ct.	5/23/81	19.1	.41					247.2	.077
		7/01/81	6.0[1]	6.00						.024
		8/27/81	17.1	<.01						.069
BC2	1981 Ct.	5/16/81	69.4	2720.53					294.5	.236
		6/29/81	60.0	.09	60.0	165.45	60.0	.09		.204
		8/17/81	28.3	6.31						.096
BC3	1981 Ct.	5/10/81	91.4	2.08					251.3	.364
		6/28/81	125.4	10.32	334.4	34069.40	125.4	10.32		.499
		8/26/81	77.1	.10						.307

[1] Catch-effort population estimate not possible. $\hat{N}_E$, $V(\hat{N}_E)$ set equal to total catch.

Appendix Table 28. The estimated abundance of the 1979 year class of coho salmon at sampling sites on Bear Creek in the period 5/80 to 8/81, and the variance of this estimate (E — catch, effort estimate, MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area.

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
BC1	1979 Co.	5/04/80	1.0[1]	1.00					247.2	.004
BC2	1979 Co.	5/06/80	11.0[1]	11.00					294.5	.037
BC3	1979 Co.	5/06/80	22.4	201.38					251.3	.089

[1] Catch-effort population estimate not possible. $\hat{N}_E$, $V(\hat{N}_E)$ set equal to total catch.

Appendix Table 29. The estimated abundance of the 1980 year class of coho salmon at sampling sites on Bear Creek in the period 5/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area.

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface Area (m^2)	Density (N/m^2)
BC1	1980 Co.	5/04/80	121.6	5.52					247.2	.492
		7/20/80	71.1	11.06						.288
		9/27/80	65.3	18.53	59.0	43.20	63.4	12.97		.256
		11/09/80	38.1	34.40	230.4	18356.57	38.5	34.34		.156
		12/21/80	42.4	.01	160.2	6146.83	42.4	.01		.172
		4/04/81	23.9	.21						.100
		5/23/81	3.0[1]	3.00						.012
BC2	1980 Co.	5/06/80	178.4	10.46					294.5	.606
		7/21/80	164.6	53.82						.559
		9/21/80	104.1	82.75	233.2	9504.64	105.2	82.04		.353
		11/08/80	54.6	6.98	103.0	4015.59	54.7	6.97		.186
		1/12/81	20.3	3.48	64.0	1876.74	20.4	3.47		.069
		3/26/80	34.2	9.27						.116
		5/16/80	18.0[1]	18.00						.061
BC3	1980 Co.	5/06/80	64.7	.03					251.3	.257
		7/22/80	47.7	2.10						.190
		9/22/80	58.2	5.48	55.0	43.47	57.9	4.87		.230
		11/07/80	69.8	1234.98	115.6	12371.17	74.0	1122.89		.294
		1/12/80	13.1	<.01	13.0	17.30	13.1	<.01		.052
		3/25/81	32.0	9.05						.127
		5/10/81	48.0[1]	48.00						.191

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[1] Catch-effort population estimate not possible. $\hat{N}_E$, $V(\hat{N}_E)$ set equal to total catch.

Appendix Table 30. The estimated abundance of the 1981 year class of coho salmon at sampling sites on Bear Creek in the period 5/80 to 8/81, and the variance of this estimate (E — catch, effort estimate; MR — mark, recapture estimate; P — pooled estimate). Density expressed in terms of low flow stream surface area.

Site	Species Age class	Date	$\hat{N}_E$	$V(\hat{N}_E)$	$\hat{N}_{MR}$	$V(\hat{N}_{MR})$	$\hat{N}_P$	$V(\hat{N}_P)$	Surface area (m^2)	Density (N/m^2)
BC1	1981 Co.	4/04/81	139.2	.40					247.2	.563
		5/16/81	98.5	2.04	167.0	1343.49	98.6	2.04		.398
		7/01/81	121.9	43.34	189.9	944.38	124.9	41.44		.505
		8/27/81	119.1	.44						.482
BC2	1981 Co.	1/12/81	8.0	.93					294.5	.027
		3/26/81	300.9	80.75						1.021
		5/16/81	163.0	24.23	198.7	387.99	165.1	22.81		.561
		6/29/81	192.2	.86	257.5	1349.66	192.2	.86		.653
		8/17/81	110.9	1.14						.377
BC3	1981 Co.	3/25/81	73.6	.31					251.3	.293
		5/10/81	87.7	.03	120.3	294.31	87.7	.03		.349
		6/28/81	88.4	47.09	84.0	21.98	85.4	14.99		.340
		8/26/81	85.3	<.01						.340

Appendix Table 31. Age composition of adult cutthroat trout captured at the downstream net.

Ocean annuli	Stream annuli	Lake annuli	Number of fish
0	1	2	1
0	1	3	1
1	1	0	5
1	1	1	3
1	2	0	7
1	2	1	2
2	1	0	17
2	2	0	14
3	1	0	16
3	2	0	6
4	1	0	3
4	1	1	1
4	2	0	5

Appendix Table 32. Kelsey Creek spawner survey data: October 9, 1979 through May 5, 1980.

	Species			
Date	Sockeye	Coho	Chinook	Cutthroat
10/09/79	0	0	0	0
10/16/79	3	0	0	0
10/23/79	0	7	0	0
10/30/79	0	10	2	0
11/06/79	0	13	1	0
11/13/79	0	4	0	0
11/20/79	0	2	0	0
11/27/79	0	7	0	0
12/05/79	0	8	0	0
12/09/79	0	14	0	0
12/17/79	0	3	0	0
12/29/79	0	8	0	0
1/04/80	0	4	0	0
1/08/80	0	0	0	0
1/16/80	0	0	0	0
2/28/80	0	0	0	5
3/05/80	0	0	0	6
3/11/80	0	0	0	9
3/19/80	0	0	0	10
3/26/80	0	0	0	7
4/02/80	0	0	0	8
4/10/80	0	0	0	11
4/16/80	0	0	0	26
4/22/80	0	0	0	9
4/30/80	0	0	0	5
5/05/80	0	0	0	0

Appendix Table 33. Kelsey Creek[1] spawner survey data, September 1980 to January 1981.

			Coho			Chinook			
Date	Visibility	Discharge (m^3/sec)	Number live	Number dead	Number new redds	Number live	Number dead	Number new redds	Comments
9/11/80	Poor	.26	0	0	0	0	0	0	Poor visibility due to bank work at 134th St.
9/24/80	Good	.28	0	0	0	0	0	0	
10/01/80	Excellent	.28	0	0	0	1	0	0	
10/10/80	Excellent	.25	0	0	0	1	2	0	
10/16/80	Fair	.26	0	2	0	1	0	0	Both coho appeared to have been speared.
10/23/80	Excellent	.25	2	0	1	4	0	0	
10/30/80	Excellent	.24	0	1	1	5	9	6	
11/12/80	Poor	.40	6	3	5	0	0	0	
11/24/80	Poor	.48	8	1	5	0	0	0	
12/05/80	Good	.94	0	1	1	0	0	0	
12/16/80	Good	.34	1	3	1	0	0	0	
1/15/81	Excellent	.35	0	0	0	0	0	0	

[1] Count area extended upstream from Kelsey Creek Park to crossing of 148th N.E.

Appendix Table 34. Kelsey Creek[1] spawner survey data, February to April 1981.

Date	Visibility	Discharge (m^3/sec)	Cutthroat Number live	 Number dead	 Number new redds	Length/sex of dead fish
2/23/81	Excellent	.51	6	2	2	♀ 473, ♂ 427
3/02/81	Excellent	.42	2	0	4	
3/09/81	Excellent	.42	1	1	3	♀ 433
3/16/81	Excellent	.65	16	4	9	♀ 460, ♀ 434, ♀ 459, ♂ 488
3/27/81	Excellent	.40	15	3	12	♀ 405
4/13/81	Excellent	.50	0	0	9	

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[1] Count area extended upstream from Kelsey Creek to crossing of 148th N.E.

Appendix Table 35. Bear Creek spawner survey data: September 24, 1979 through April 28, 1980.

	Species			
Date	Sockeye	Coho	Chinook	Cutthroat
9/24/79	225	0	0	0
10/02/79	329	0	0	0
10/09/79	667	11	0	0
10/17/79	978	56	0	0
10/27/79	535	209	4	0
11/02/79	318	227	5	0
11/10/79	89	83	1	0
11/17/79	21	79	0	0
11/25/79	19	75	0	0
12/02/79	10	78	1	0
12/09/79	9	51	0	0
12/18/79	2	58	0	0
12/27/79	3	84	0	0
1/06/80	0	43	0	0
1/13/80	0	21	0	0
1/20/80	0	8	0	0
3/01/80	0	3	0	0
3/08/80	0	5	0	3
3/15/80	0	0	0	2
3/30/80	0	0	0	0
4/06/80	0	0	0	0
4/20/80	0	0	0	3
4/28/80	0	0	0	0

Appendix Table 36. Bear Creek spawner survey data, September to December, 1980.

			Sockeye[1]			Coho[2]			
Date	Visibility	Discharge (m^3/sec)	Number live	Number dead	Number new redds	Number live	Number dead	Number new redds	Comments
9/11/80	Excellent	.25	14	0	0	0	0	0	
9/18/80	Excellent	.26	127	1	1	1	0	0	
9/26/80	Excellent	.31	353	55	41	7	0	1	2 chinook, 1 chinook redd
10/03/80	Excellent	.31	239	82	17	28	0	2	5 chinook, 1 chinook redd
10/09/80	Excellent	.25	72	46	13	55	6	4	2 dead chinook
10/17/80	Excellent	.25	1	25	7	56	16	7	
10/24/80	Excellent	.24	0	1	0	61	21	13	1 chinook
11/08/80	Fair	.85	0	0	0	75	11	18	Stream high, but clear
11/14/80	Good	.37	0	0	0	11	13	5	2 chinook
12/11/80	Excellent	.74	0	0	0	9	3	9	

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[1] Count area extended upstream from foot of N.E. 148th to approximately .25 miles upstream.

[2] Count area extended upstream from foot of N.E. 148th to the crossing of the Woodinville-Duvall Road.

Appendix Table 37. Bear Creek[1] spawner survey data, February to March 1981.

			Cutthroat			Steelhead		
Date	Visibility	Discharge (m^3/sec)	Number live	Number dead	Number new redds	Number live	Number dead	Number new redds
2/27/81	Good	.99	1	0	0	0	0	0
3/18/81	Excellent	.48	0	0	0	5	0	1
3/27/81	Excellent	.51	0	0	0	4	0	2

[1] Count area extended upstream from foot of N.E. 148th to the crossing of the Woodinville-Duvall Road.

Appendix Table 38. Estimated survival of sockeye salmon fry from release at the streamside incubation boxes (located at Kelsey Creek Park or on Valley Creek) to recovery at the downstream net.

	Number released			
Date	Kelsey	Valley	Number recovered	Estimated survival
3/16/81	294	0	39	.40
3/31/81	6	4391	80	.06
4/02/81	1	1940	76	.12
4/06/81	0	713	20	.08
4/09/81	0	762	3	.01